End-of-Life in South African Product Life Cycle Assessment

Submitted to the University of Cape Town in fulfilment of the requirements for the degree of Masters of Science in Engineering

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December 2017

STATEMENT OF ORIGINALITY

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ABSTRACT

Life cycle assessment (LCA) is a tool specifically developed for quantifying and assessing the environmental burden of a product across its entire life cycle, thus providing powerful support for sustainable product design. There exists a geographical imbalance in the adoption and distribution of LCA studies, with a notably poor penetration into developing countries, resulting from a lack of technical expertise, reliable data, and an inability to engage with the key issues of developing countries. These challenges are particularly prevalent in waste management.

The limitations in current LCA capacity for representing product end-of-life, coupled to the disparity in waste management practices between developed and developing countries means that LCA is currently unable to accurately model product end-of-life in South Africa. This means that, for imported products designed on the basis of LCA, the upstream impacts may be accurate, while the end-of-life is not. Therefore, to improve the use of LCA as a tool to support sustainable product design, there is a need to develop life cycle datasets and methods that accurately reflect the realities of waste management in developing countries. The objectives of this dissertation are to (i) identify the current shortcomings of existing LCA datasets in representing the end-of-life stage of general waste in a South African context, and (ii) propose modifications to existing datasets to better reflect the realities of waste management in a South African context and extract lessons from this for use elsewhere.

To meet these objectives, research was undertaken in three main stages, with the outcome of each stage used to inform the development of each subsequent stage. The first stage aimed to establish the status quo with regards to general waste management in South Africa. This investigation was informed through a desktop review of government and other publicly available reports, supplemented by field work and stakeholder engagements. These results formed the basis for the second stage: a review of LCA capacity for representing product end-of-life in the South African context. The review of datasets was limited to those contained within SimaPro v8.3 and was undertaken with the aim of understanding the extent to which current datasets are capable of representing South African waste management practices. Finally, three cases of existing LCA datasets were explored. This included testing modifications that could be made in an attempt to improve their applicability to the South African reality.

In South Africa, a major limitation in developing a quantified mapping of waste flows lies in the paucity of reliable waste data and the exclusion of the contribution of the informal sector in existing waste data repositories. It was estimated that South Africa generates approximately 12.7 million tonnes of domestic waste per annum, of which an estimated 29% is not collected or treated via formal management options. For both formal and informal general waste, disposal to land (landfill and dumping) represents the most utilised waste management option. Landfill conditions in South Africa range from well-managed sanitary landfills to open dumps. Considering only licensed landfill facilities, it is estimated that large and medium landfill sites accept the majority of South Africa's general waste (54% and 31% respectively), while the balance is managed in small (12%) and communal (3%) sites. Considering the quantity of informal domestic waste enables a crude estimation of household waste distribution between different landfill classes. In this instance, while the majority of waste (40%) is still managed in large formal landfill sites, an appreciable quantity (26%) is managed in private dumps.

Within SimaPro v8.3 landfill disposal is best represented by the sanitary landfill datasets contained within the ecoinvent v3.3 database. SimaPro preserves the modular construction of the ecoinvent dataset, meaning that various generic modifications to these datasets can be made, such as the elimination or addition of burdens, redefinition of the value of a burden, or substitution of a linked dataset. Practically, such modifications are limited to process-specific burdens. However, waste-

specific burdens are of greater significance in the life cycle impact assessment (LCIA) result of a landfill process. Waste-specific emissions are generated using the underlying ecoinvent landfill emission model. The current model structure allows for the parametrisation of waste composition in addition to landfill gas (LFG) capture and utilisation efficiencies. However, besides the incorporation of a methane correction factor to account for the effect that various site conditions have on the waste degradation environment, the extent to which the existing model can be adapted to represent alternative landfill conditions is limited. This is particularly true in the case of leachate generation and release. Although adaptation that incorporates the effect of climatic conditions on waste degradability and emission release is possible, this requires a high level of country-specific data and modelling expertise. Thus, the practicality of such a modification within the skills set of most LCA practitioners is questionable. Further limitations in the existing modelling framework include its inability to quantify the potential impacts of practices characteristic of unmanaged sites such as open-burning, waste scavenging, and the presence of vermin and other animal vectors for disease.

Analysis of the LCIA results for different landfill scenarios showed that regardless of either the deposited material or the specific landfill conditions modelled, the time frame considered had the most pronounced effect on the normalised potential impacts. Regardless of landfill conditions, when long-term leachate emissions are considered, freshwater and marine ecotoxicity impacts dominate the overall potential impacts of the site. This result implies that if landfill disposal is modelled over the long-term, the potential impacts of the process has less to do with site-specific conditions than it does to do with the intrinsic properties of the material itself. Given the ensuing extent of degradation that occurs over the time frame considered, the practise of very long-term modelling can equalise landfills that differ strongly in the short-term. In terms of product design on the basis of LCA, the choice of material can be more strongly influenced by the time frame considered than the specific landfill scenario.

From a short-term perspective, for fast degrading materials the impacts incurred from leachate emissions and their subsequent treatment are of lesser importance than those arising from LFG. From a long-term perspective by contrast, leachate emissions have a significant effect on the LCIA result. Investigation into the effect of reduced precipitation on the LCIA result showed that the exclusion of leachate emissions lowers the potential impacts of a number of impact categories, with the most substantial quantified reduction observed in the freshwater and marine ecotoxicity impact categories. This result implies that for dry climates, the long-term impacts of landfilling could be significantly lower than when compared to landfill under temperate conditions, with the potential impacts of the waste remaining locked-up in the landfill.

Given quantified findings on South Africa's dependence on both formal and informal disposal, and the variation in landfill conditions across the country, it can be concluded that LCA results for the impacts of products originating from global supply chains, but consumed and disposed of in South Africa, will be inaccurate for the end-of-life stage if modifications to end-of-life modelling are not made. The findings from this dissertation provide the basis for i) a crude estimate of 'market shares' of different disposal practises and ii) guidelines for parameterisation of material specific emission factors, in particular for shorter term emissions, focused on LFG and leachate emissions.

ACKNOWLEDGEMENTS

I would like to express my gratitude to my supervisor, Prof. Harro von Blottnitz, for his insight, guidance and support throughout the duration of the project.

I would also like to extend my thanks to my co-supervisor, Dr. Philippa Notten, who has provided me with endless opportunities and insights and has been a constant pillar of support.

The financial support of the *National Research Foundation's Global Change, Society and Sustainability Research Program (GCSSRP)* is gratefully acknowledged.

I would like to acknowledge the time and insight of all of the stakeholders who I engaged with over the course of this project. Special thanks are extended to Eddie Hanekom from the Western Cape Department of Environmental Affairs and Development Planning for his willingness to share the Western Cape landfill audit report data and significant insight into waste management at both a provincial and national level.

Finally, I wish to thank my friends and family for their interest in this work and their on-going encouragement and support.

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Table C.6 Inventory values for polyethylene, cardboard and compostable material waste as specified in the ecoinvent incineration model........ C-6

LIST OF ACRONYMS

CDM	Clean Development Mechanism
CH	Switzerland
DEA	Department of Environmental Affairs
DEAT	Department of Environmental Affairs and Tourism
DOC	Degradable Organic Carbon
DWAF	Department of Water Affairs and Forestry
EPA	Environmental Protection Agency
EPR	Extended Producer Responsibility
GDP	Gross Domestic Product
GHG	Greenhouse Gas
GLO	Global
HDPE	High Density Polyethylene
IDPs	Integrated Development Plans
IndWMP	Industry Waste Management Plan
IPCC	Intergovernmental Panel on Climate Change
IRMA	Information Resources Management Association
ISO	International Organisation for Standardization
IWMP	Integrated Waste Management Plan
LCA	Life Cycle Assessment
LCI	Life Cycle Inventory
LCIA	Life Cycle Impact Assessment
LCM	Life Cycle Management
LFG	Landfill Gas
MCF	Methane Correction Factor
MECs	Members of Provincial Executive Committees
MRD	Maximum Rate of Deposition
MSW	Municipal Solid Waste
NEM:WA	National Environmental Management: Waste Act, No. 59 of 2008
NPSWM	National Environmental Management: Waste Act, No. 59 of 2008. National Pricing Strategy for Waste Management
NWBR	National Waste Baseline Report

NWMS	National Environmental Management: Waste Act, No. 59 of 2008. National Waste Management Strategy
OECD	Organisation for Economic Co-Operation and Development
PETCO	PET Recycling Company
POLYCO	Polyolefin Recycling Company
PRASA	Paper Recycling Association of South Africa
PROs	Producer Responsibility Organisation
PSPC	Polystyrene Packaging Council
RoW	Rest of World
RSA	Republic of South Africa
S	System Process Record
SAVA	South African Vinyls Association
SAWIC	South African Waste Information Centre
SAWIS	South African Waste Information System
SIC	Standard Industrial Classification
TK	Transfer Coefficient
U	Unit Process Record
UCT	University of Cape Town
WCDEADP	Western Cape Government Department of Environmental Affairs and Development Planning
WCED	World Commission on Environment and Development
WDF	Waste Disposal Facility
WGRs	Waste Generation Rates
WHO	World Health Organisation
WIS	Waste Information System
WWTP	Wastewater Treatment Plant

LIST OF SYMBOLS

Chemical Compounds

DB eq	Dichlorobenzene equivalents
CH_4	Methane
CO	Carbon monoxide
CO_2	Carbon dioxide
CO_2 eq	Carbon dioxide equivalents
H_2S	Hydrogen sulfide
N_2O	Nitrogen dioxide
NMVOC	Non-methane volatile organic compound
NO_x	Nitrogen oxide
VOC	Volatile organic compound
XOC	Xenobiotic organic compounds

Landfill Size Classification

B^-	Sporadic leachate generation likely
B^+	Potential for significant leachate generation
C	Communal landfill (MRD < 25 tonnes/day)
G	Landfill accepting general waste
L	Large landfill (MRD> 500 tonnes/day)
M	Medium landfill (MRD >150, <500 tonnes/day)
S	Small landfill (MRD > 25, <150 tonnes/day)

Mathematical and Modelling Symbols

%	Percentage
$\%gas_e$	Fraction of the released amount of element *e* emitted in landfill gas [weight %]
$B^0_{i,x}$	Burden for exchange *i* from 1 kg pollutant *x* in 1 m^3 of wastewater
B_i	Total burden for exchange *i* from leachate treatment
$B_{i,W}$	Burden for exchange *i* from 1 m^3 of unpolluted wastewater
$B_{i,x}$	Burden for exchange *i* from pollutant *x* in leachate
D	Decomposition rate of waste [kg/kg in 100 years]
$D_{Leachate}$	Dissimilation coefficient for leachate
D_{LFG}	Dissimilation coefficient for landfill gas
DOC_f	Degradability factor

e	Chemical element
$E_{short\text{-}term,gas,e}$	Short-term emission of the element *e* to landfill gas [kg/kg waste]
$E_{short\text{-}term,leach,e}$	Short-term emission of the element *e* to leachate [kg/kg waste]
L0	Ultimate convertible amount of decay
m_e	Concentration of element *e* in waste fraction [kg/kg waste]
m_x	Mass of pollutant *x* in leachate 0 – 100 years [kg]
r_e	Average release factor for element *e* [kg/kg]
TK	Transfer coefficient
$TK_{short\text{-}term,gas,e}$	Short-term transfer coefficient of the element e to landfill gas [kg/kg] element
$TK_{short\text{-}term,leach,e}$	Short-term transfer coefficient of the element e to leachate [kg/kg] element
V	Mean annual leachate output from the landfill per kg of waste [m^3/a per kg waste]

Units of Measurement

kg	Kilogram
kWh	Kilowatt hours
L	Litres
m	Metres
m^2	Square metres
m^3	Cubic metres
Mt	Megatonne (one million metric tonnes)
mPe	Normalised environmental impact potential (as a person equivalence: impact potential per person per year*1000)
p	Demand per kilogram of waste. Reported as a fraction of the demand associated with the construction of a 1.8 million m^3 landfill for untreated municipal waste
Pe	Normalised environmental impact potential (as a person equivalence: impact potential per person per year)

Chapter 1

INTRODUCTION

Patterns of human development have been officially recognised since at least 1987 to be unsustainable in light of increasing population, growth in material consumption, and the finite carrying capacity of the earth (World Commission on Environment and Development [WCED], 1987). Given the increasing pressure this places on already constrained resources, there is a growing need to redirect current consumption and production patterns in a manner consistent with the principles of sustainable development (Meadowcroft, 2007). The global discrepancy in the level of waste management structures and practices employed in different countries suggests that there might not be a single generic design solution to facilitate product sustainability at its end-of-life, thus necessitating the evaluation of a design decision within the local context.

1.1 Background

1.1.1 Overview of Sustainable Development

The concept of sustainable development has become a focal point of global governance, and is rooted in the report of the World Commission on Environment and Development (WCED), which defines sustainable development as "paths of human progress that meet the needs and aspirations of the present generation without compromising the ability of future generations to meet their needs" (WCED, 1987:43). The so called *Brundtland Report* further emphasises the need to curb historically unsustainable yet ongoing development patterns to address — and divert possible increases in — severe global economic and social disparities and environmental threats (WCED, 1987). Over the past three decades, the concept of sustainable development — of which numerous other definitions have been proposed — has emerged with increasing prominence as the caveat issued by the Brundtland Report is heeded: current patterns of development and consumption cannot be sustained amidst increasing population, growth in human consumption, and the finite carrying capacity of the earth (Charter & Tischner, 2001).

It has been suggested that the emergence of sustainable development as a concept invoking widespread support and interest illustrates an important shift in understanding the relationship that exists between humanity and nature (Hopwood, Mellor & O'Brien, 2005). In contrast to historical development patterns, in which environmental issues were separated from socio-economic concerns, sustainable development recognises the interplay between environmental, social and economic issues (Hopwood, Mellor & O'Brien, 2005). As an illustration of this interplay, sustainable development can be regarded as a "nested" concept in which the economy lies within society, which is in turn nested within the environment (Giddings, Hopwood & O'Brien, 2002:191). This is illustrated in Figure 1.1 overleaf.

The concept of sustainable development provides a means to redirect the human development trajectory to achieve a balance between all three development spheres (Meadowcroft, 2007). However, the tendency of this concept to be viewed as a "pathway to all that is good and desirable in society" (Holden, Linnerud & Banister, 2014:130) has been criticised as supporting the proliferation of local factors and indices that increase the complexity of the concept and reduce its use in policymaking (Holden, Linnerud & Banister, 2014). Despite this criticism, sustainable development remains high on both the international and national agenda.

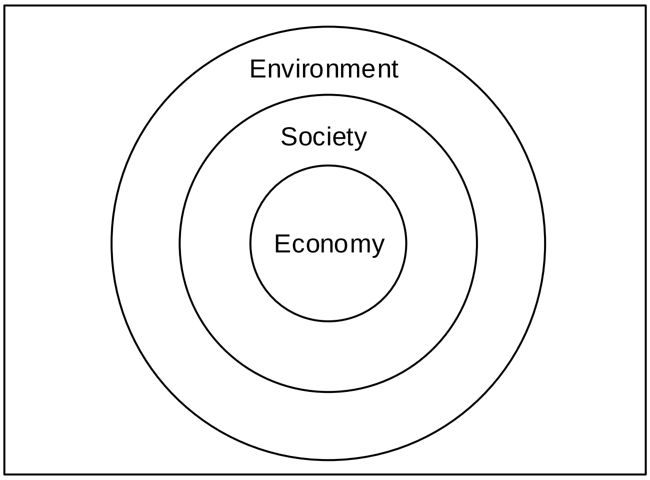

Figure 1.1 Schematic representation of the interplay between society, environment, and economy in supporting sustainable development as a nested concept (Adapted from Giddings, Hopwood & O'Brien, 2002:192)

Following the outcome of the United Nations Conference on Sustainable Development, held in Rio de Janeiro in 2012, a set of sustainable development goals was defined in 2015 as part of the United Nations Sustainable Development Agenda, to which 193 countries (including South Africa) are signatories. This agenda defines 17 Sustainable Development Goals across the three major spheres of sustainability and a set of corresponding targets (United Nations [UN], 2015) with the mandate to "end poverty, protect the planet, and ensure prosperity for all" (UN, 2017). The Sustainable Development Goals build on the Millennium Development Goals — a set of time-bound targets agreed upon at the Millennium Summit in 2000 to address basic human rights to health, education, shelter and security by 2015 — and recognise the need for more decisive action with a more ambitious and directed agenda (United Nations Development Programme [UNDP], 2015). The Sustainable Development Goals also recognise and set targets to address specific issues not included in the Millennium Development Goals, such as climate change and sustainable consumption and production (UNDP, 2015).

The inclusion of a directed development goal to promote the adoption of sustainable consumption and production patterns (Goal 12 of the United Nations Sustainable Development Agenda) aims to address the increasing global demand for constrained resources. The targets of this Goal are intended to change the manner in which resources and goods are consumed and produced by promoting efficient resource management, improved efficiency in both production and supply chains, and responsible waste management and disposal (UNDP, 2017). Specific emphasis has been placed on improving the sustainability of waste management, with directed targets aimed at encouraging the adoption of sustainable practices at company level and actively seeking to reduce waste generation through prevention, reduction, recycling, and re-use (UNDP, 2017).

The realisation of Goal 12 requires global commitment to meeting the proposed targets. To this end, one of the focus areas of this Goal is to address the disparity in consumption and production that occurs between developed and developing nations. It is one of the objectives defined for this Goal to improve the capacity for sustainability in developing countries by strengthening both scientific and technological capacity (UNDP, 2017).

1.1.2 Incorporating the Principles of Sustainable Development into Product Design

Conventionally, product design strategies are directed towards meeting technical, aesthetic and cost targets with limited consideration for the post-consumer use stages of a product's life (Howarth & Hadfield, 2006). Traditionally, product design has been based on cost/profit models, aimed at minimising design and manufacturing costs to maximise profits (Kaebernick, Kara & Sun, 2003). However, international directives and increasingly stringent environmental policies and law are placing pressure on manufacturing industries to develop environmentally sustainable products (Maxwell & van der Vorst, 2003), thus requiring consideration of the impacts of a product beyond the point of sale. It has been suggested that the development of sustainable products requires that the principles of sustainable development be incorporated into the design process (Howarth & Hadfield, 2006).

While the Sustainable Development Goals were formally adopted in 2015, prior to this, notable inroads had already been made into promoting sustainable production and consumption patterns, with sustainable product design (or sometimes equivalently "design for the environment") widely considered a key practice for sustainable development (Chen et al., 2012:348). Indeed, before the formalisation of the United Nations sustainability targets, Chen et al. (2012) notes that a number of directives and regulations aimed at encouraging sustainable design practices were considered and imposed by various governments in response to factors including increasingly stringent environmental legislation and strong public interest in sustainable purchasing.

Achieving a sustainable product design cannot, however, be considered in isolation, as "good value" remains an important criterion in consumer purchasing decisions (Chen et al., 2012:349). Thus, good design requires the integration of good environmental performance with the technical characteristics of the product's design (Fargnoli & Kimura, 2006). Indeed, it has been suggested that the design procedure should be considered a "cross-functional activity" (Fargnoli & Kimura, 2006:190), balancing the demands and expectations of consumers, the needs of the company, legal and regulatory requirements, and the needs of society across the life cycle of the product (Fargnoli & Kimura, 2006). The integration of life cycle thinking (LCT) into traditional design approaches has been recognised as an approach capable of supporting a balance between social, economic, and environmental considerations (Fargnoli & Kimura, 2006).

1.1.3 Life Cycle Thinking in Product Design

LCT extends the traditional scope for consideration from the production and manufacture of a product to include the environmental, social, and economic impacts of the product over its entire life cycle. The use of LCT as a design tool is promoted by the Life Cycle Initiative, established in 2002 by a partnership of the United Nations Environment Program (UNEP) and the Society for Environmental Toxicology and Chemistry (SETAC).

The UNEP/SETAC Life Cycle Initiative has three objectives (United Nations Environment Program [UNEP] & Society of Environmental Toxicology and Chemistry [SETAC], 2011:2):

1. *Enhance the global consensus and relevance of existing and emerging life cycle approaches and methodologies.*
2. *Facilitate the use of such approaches worldwide by encouraging life cycle thinking in decision-making for enterprises, public authorities, and consumers.*
3. *Expand capability worldwide by applying and improving life cycle approaches.*

In response to this initiative, the life cycle management (LCM) of products has become a regulatory instrument in many countries, informed by the practise of Life Cycle Assessment (LCA) (UNEP & SETAC, 2012). However, it has been asserted that there exists a geographical imbalance in the adoption and distribution of LCA studies, with the majority concentrated in Europe and America with poor penetration into developing countries (Laurent et al., 2014a). The penetration of LCA into developing countries is typically hampered by a lack of technical expertise, reliable data, and the inability of LCA to engage in the key issues of developing countries (UNEP & SETAC, 2009).

Furthermore, the nature of the globalised economy can present a challenge to LCA methodology given that product value chains are typically spread across countries around the globe (UNEP & SETAC, 2012). Given this globalised economy, product designers are increasingly tasked with designing sustainable products that are to be distributed and consumed in an unequal society. It is the role of the LCA practitioner to develop the quantitative basis that informs design decisions and thus, with methodological restrictions on the conditions that are modelled, decisions leading to sustainable outcomes in one country can have unexpected consequences in another (UNEP & SETAC, 2011).

In order to address the geographical imbalances and challenges in the application of LCA methodology towards sustainable product development, key steps have been identified towards promoting the advancement of the UNEP/SETAC Life Cycle Initiative, particularly into developing countries. The key steps include developing technical expertise in developing countries and emerging economies where there is a lack of local capabilities, the acquisition of representative local data, and strengthening the application of LCA through consideration of geographical trade-offs and local practices to provide improved decision support regarding product sustainability across the globe (UNEP & SETAC, 2011).

1.1.4 Application of Life Cycle Thinking to Product End-of-Life

Within the European Union in particular, product-orientated legislation regarding the sustainability of product design from a LCT perspective is largely based on the principle of Extended Producer Responsibility (EPR). The objective of EPR is to assign “long-term environmental responsibility of products to producers” (Sachs, 2006:53), hence directing the physical and financial responsibility for the environmental impacts of waste towards the generators of waste (Nahman, 2010). The EPR principle is intended to internalise costs associated with downstream product management, thus promoting the conversion of the linear “cradle-to-grave” production and distribution patterns into a “cradle-to-cradle” system, encouraging product designers to incorporate sustainable practices such as waste reduction, recycling and re-use into their product design (Sachs, 2006:53).

Extending the responsibility of the producer beyond the manufacture of the product has important implications for the product’s end-of-life. The reduction in waste generation through promotion of the “three Rs” (UNEP, 2011:294) is aligned with the principles laid out in the waste hierarchy (shown in Figure 1.2). The waste hierarchy represents the internationally accepted approach towards sustainable solid waste management, supporting a shift away from traditional end-of-pipe treatment, such an landfilling, and instead promoting resource efficiency (UNEP, 2011).

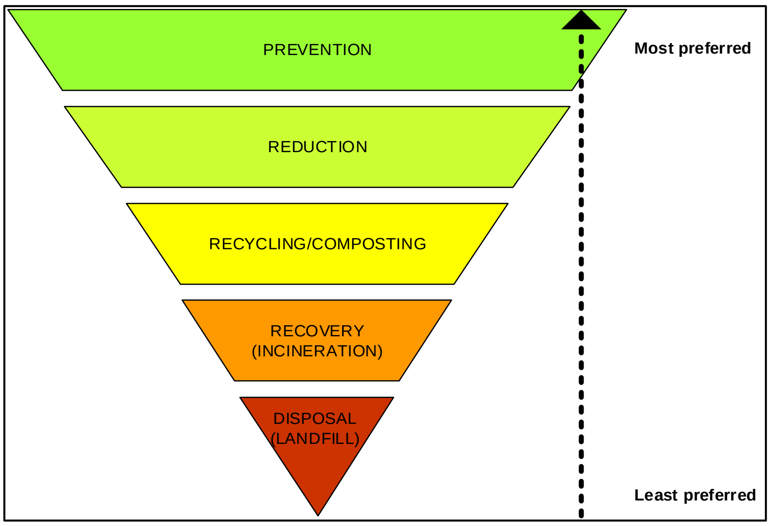

Figure 1.2 The waste hierarchy (Adapted from UNEP, 2011:294)

Global disparities in wealth and economic and industrial development affect the level of waste services and management practices between and within different countries (Hoornweg & Bhada-Tata, 2012). Where developed countries are commonly characterised by a complete and highly effective waste management service, with increasing emphasis on waste diversion and material/energy recovery (Hoornweg & Bhada-Tata, 2012), it has been suggested by Onu (2001) (as cited by Matete and Trois (2008:1480)) that developing countries, by contrast, reflect the following characteristics:

- Ineffective or partial waste collection practices and/or lack of environmental control systems
- Illegal dumping, littering, and waste scavenging
- Limited environmental and waste awareness of the general public

The difference in waste management between developing countries and countries where LCA methods are well established implies that LCA results for products sourced from a global supply chain, but sold and disposed of in developing countries, might be accurate for describing the impacts associated with the upstream part of the life cycle, but inaccurate for the end-of-life stage. The extent to which alternative end-of-life scenarios can be evaluated (so as to assess the sustainability of a product in a broader geographical context) is limited, as the challenges associated with the uptake of LCA in developing countries, such as limited data and its inability to engage with the key issues of developing countries, are particularly prevalent with regards to waste management (UNEP & SETAC, 2011).

1.2 Problem Statement

Adopting a life cycle thinking approach towards product design has been recognised as a means to reduce the environmental impacts of the billions of products made, sold, consumed, and disposed globally every day. Consequently, the life cycle management of products has become a regulatory instrument in many countries, informed by LCA. Improving the sustainability of the end-of-life stage in particular has received much emphasis, with products increasingly designed in alignment with the principles of the waste hierarchy (Figure 1.2), e.g. designing for recyclability to promote resource efficiency. However, there is a disparity in waste management practises occurring between developed and developing countries, with ineffective waste management common across much of the developing world. Furthermore, limitations in data availability and the inability of current LCA capacity to engage with the key issues of developing countries, amongst else, hampers the ability of LCA to represent the end-of-life fate of products in many countries accurately.

Therefore, the LCA results for the impacts of products originating from global supply chains but consumed and disposed of in South Africa might be accurate for the upstream part of the life cycle, but inaccurate for the end-of-life stage. The reason for this is that waste management practices in South Africa, a developing country, may be different from practices in countries where life cycle methods are well established. It is therefore necessary to investigate the extent to which current LCA capacity can be used to represent waste management in South Africa, and where short-comings are identified, develop life cycle datasets and methods that accurately reflect the realities of waste management in a South African context. This is necessary to enable decision-making that supports improved sustainability across a product's life cycle, regardless of the geographical context.

1.3 Dissertation Objectives and Research Approach

To improve the use of LCA as a tool to support sustainable product design, it is necessary to develop life cycle datasets and methods that accurately reflect the realities of waste management in many countries. This dissertation aims to do so in a South African context and to extract lessons from this for use elsewhere. The objectives of this dissertation are therefore as follows:

1. Identify the current shortcomings of existing LCA datasets in accurately representing the end-of-life stage of general (non-hazardous) waste in a South African context.
2. Propose modifications to existing datasets to better reflect the realities of waste management in a South African context, ultimately to enable better decision-making by product designers and brand owners, for better environmental outcomes.

This study is undertaken in three main stages, with the outcome of each stage used to inform the development of subsequent stages. An overview of each stage is provided below.

Stage 1: Establish the Status Quo for General Waste Management in South Africa

The first stage of the research aims to establish the status quo with regards to the management of general waste in South Africa through an investigation into the South African waste sector. This investigation is primarily informed by a desktop review of government and other publicly available reports and supplemented by a field work component in which stakeholders within the waste management industry were interviewed and various waste management facilities visited.

Stage 2: Review of Current LCA Capabilities for Representing Product End-of-Life in South Africa

The findings from Stage 1 provide the basis for the second stage of the research: a review of LCA capacity for representing product end-of-life in a South African context. The review of datasets is limited to those contained within one widely used LCA software. The aim of the review is to ascertain the extent to which current LCA datasets are capable of representing South African waste management practices and investigate the possible adaptations that can be made to existing datasets to improve their application to local conditions.

Stage 3: Approximate Product End-of-Life in South Africa by Means of Dataset Modification

The final stage represents a synthesis of the findings from the first and second stage of the research. The aim of this stage is to develop scenarios to represent South African waste management practices by means of imposing the modifications identified in Stage 2 to existing SimaPro LCA datasets. The effect of these modifications on the LCIA result for the waste practice is investigated. The environmental performance of each scenario is assessed for three different materials.

1.4 Scope

Various discrepancies exist in waste management between developing countries and developed countries were LCA methods are typically well established. Therefore, products designed on the basis of LCA for the best environmental outcomes in one context, do not necessarily translate as such in South Africa. This research is driven by the importance of holding companies and the state to account for the environmental burdens of their waste, hence enabling environmentally conscious design, which lies in support of the principles of sustainable development.

This dissertation investigates the representation of the end-of-life stage within the broader context of a product LCA undertaken or commissioned by a product developer or brand manager. It is therefore not the express intention of this study to develop detailed end-of-life datasets to represent waste management in South Africa, but rather to identify practical adaptations that could feasibly be undertaken by a product developer or LCA practitioner in lieu of representative end-of-life datasets for local conditions. Although waste management practices occurring in South Africa, for which there is no appropriate or corresponding model available, may be identified through this research, the development of such models for use in life cycle impact assessment is beyond the scope of this study.

Product designers work with a wide variety of materials, of which the majority become so-called general waste at the end-of-life, with small amounts designated as hazardous waste. This study focuses on general waste with particular emphasis on municipal solid waste (MSW). MSW is in itself a fairly broad categorisation and the definition of MSW constituents can vary. This can introduce uncertainties when quantifying and/or comparing waste estimates. For this reason, although MSW forms the basis for the investigation, on occasion a more specific focus on household waste is adopted.

1.5 Dissertation Structure

The structure of this dissertation is presented schematically in Figure 1.3. The introduction to the dissertation (presented in this chapter) is proceeded by a review of relevant literature, focusing on understanding global end-of-life management practices and how they are represented within LCA methodology. Chapter 3 presents the overarching research questions, key questions, and methodological approach used to meet the objectives defined for this dissertation.

Based on the nature of the research questions posed, the research was undertaken in three distinct stages, as explained in Section 1.3 above. Research into these stages was undertaken sequentially, with the outcomes of each stage used to inform the development of each subsequent stage. The results and discussion for each stage are presented sequentially in Chapter 4, Chapter 5, and Chapter 6. The dependence of the methodological development on the results of previous stages is illustrated with the two-way arrow shown in Figure 1.3. Chapter 7 provides a consolidation of the results and discussion presented in Chapter 4 – Chapter 6. This chapter highlights the key findings arising from this study, presents conclusions reflecting on the study as a whole and provides recommendations for future work.

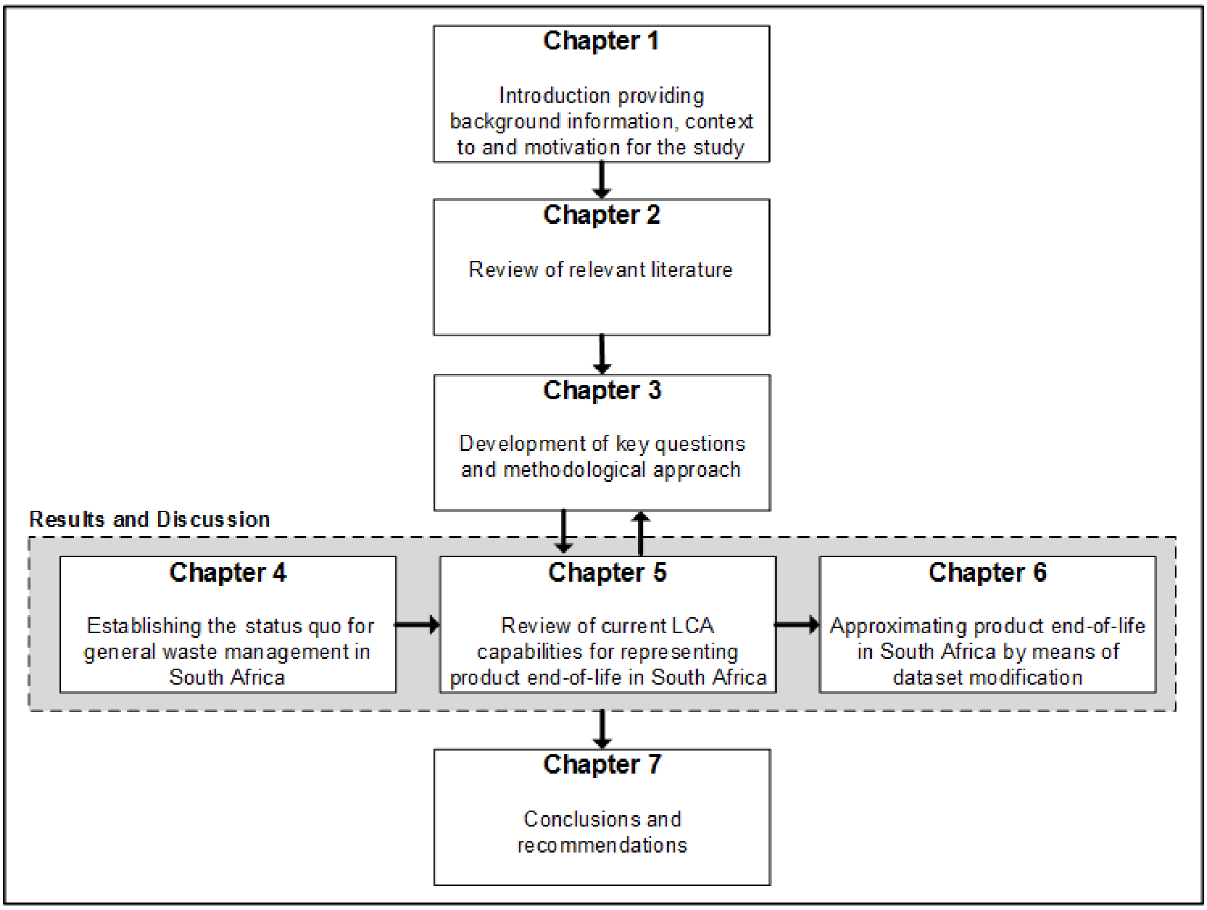

Figure 1.3 Schematic representation of the dissertation structure

Chapter 2

LITERATURE REVIEW

The aim of this chapter is to review the status quo regarding end-of-life modelling in LCA, with particular emphasis on its application to, and uptake in, developing countries. The chapter starts with an overview of LCA methodology in general, from which focus is then directed towards the end-of-life stage. Thereafter, a review of global municipal waste management is undertaken so as to identify common end-of-life management options for products and ascertain the extent to which waste management practices differ across the world. This is followed by an analysis of landfill modelling, with the aim of identifying the various challenges and limitations to the representation of the landfill process in LCA.

2.1 Environmental LCA

2.1.1 Overview of the LCA Approach

LCA is a tool that was specifically developed for quantifying and assessing the overall environmental burden of a product across its entire life cycle, thus providing a powerful support for sustainable product design. In a LCA, the environmental impacts ensuing from all stages of the life cycle of a product or service are considered, from raw material extraction for production through to disposal and waste management, thus providing a “cradle” to “grave” analysis (Baumann & Tillman, 2004:19). The advantage of such an analysis, which studies the entire product system, is that it avoids the “sub-optimisation” that may arise from an analysis in which not all processes are considered (Baumann & Tillman, 2004:21). The outcomes of a LCA are typically directed towards evaluating the impact of a product or service in three generic areas of protection, namely human health, the ecosystem, and natural resources (Ciambrone, 1997). These outcomes can therefore facilitate the manufacture, design, and management of more environmentally benign products (Ciambrone, 1997).

Increasing interest into and application of LCA since its inception has promoted the development of an international standard for LCA (Baumann & Tillman, 2004). The objective of this standard is to frame the requirements for conducting a LCA, “while leaving the actual mechanics of analysis — data collection, normalisation, calculation, interpretation, etc. — to the practitioner” (Pryshlakivsky & Searcy, 2013:115). The latest standard, ISO 14044:2006 (International Organisation for Standardization [ISO], 2006), provides a revised and updated set of standards that cancel and replace those outlined in earlier publications (ISO, 2006). According to this standard, the purposes of a LCA are as follows:

- Identifying opportunities to improve the environmental performance of products at various points in their life cycle
- Informing decision-makers in industry, government or non-government organizations (e.g. for strategic planning, priority setting, product or process design or redesign)
- Selection of relevant indicators of environmental performance, including measurement techniques
- Marketing (e.g. implementing an eco-labelling scheme, making an environmental claim, or producing an environmental product declaration)

2.1.2 Overview of LCA Methodology

Although LCA methodology is not yet considered exhaustive (Ling-Chin, Heidrich & Roskilly, 2016), it is well developed, and with ISO 14044:2006 (ISO, 2006) it provides a technique for LCA that operates within a systemised framework (Pryshlakivsky & Searcy, 2013). LCA methodology can be divided into four major phases, shown in Table 2.1.

Table 2.1 Overview of the major components of LCA methodology (Adapted from ISO 14044:2006, 2006)

Phase	Objective
1. Goal and Scope Definition	Definition of subject and intended use of study. This informs the scope of the study in terms of the system boundary definition and level of detail.
2. Life Cycle Inventory (LCI) Analysis	Inventory of input/output data with regards to the system being studied, involving the collection of the data necessary to meet the goals of the defined study.
3. Life Cycle Impact Assessment (LCIA)	Provides additional information and impact modelling to help assess a product system's LCI results to better understand the environmental significance thereof.
4. Interpretation	Summary and discussion of the results of the LCI or LCIA. These results are discussed as a basis for conclusions, recommendations, and decision-making, in accordance with the goal and scope definition

Although the objectives outlined for each phase can be broadly summarised as shown in Table 2.1, in practice these four phases support the framework for the full analysis, which contains a variety of different elements and components under each phase. Although the LCA methodology can be presented as distinct phases, the LCA procedure is iterative, resulting in the interaction between different phases throughout the procedure (Baumann & Tillman, 2004). The framework for LCA is therefore commonly presented schematically, enabling the visualisation of the iterative nature of LCA. A widely used schematic representation of this framework is shown in Figure 2.1.

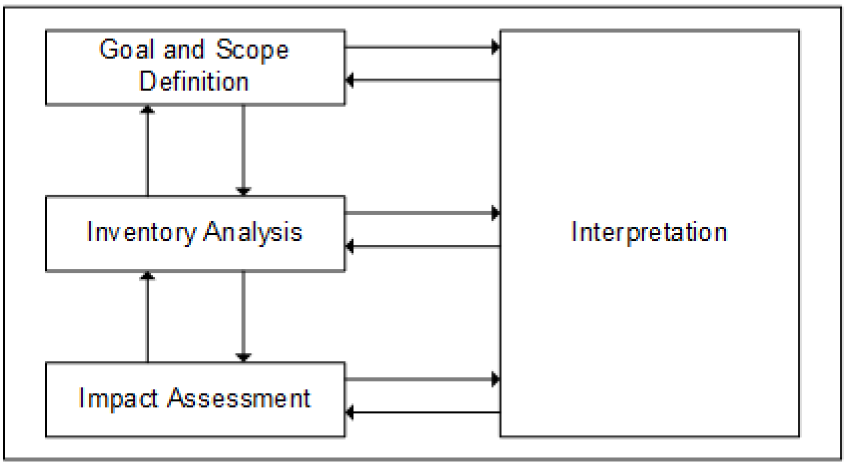

Figure 2.1 Schematic illustration of the LCA framework (Adapted from Baumann & Tillman, 2004:20)

The application of the LCA framework to a cradle to grave analysis of a product or system requires the development of a life cycle model. The life cycle model is used to represent the system under study and determine the quantitative inputs and outputs to be considered in the LCA. Figure 2.2 overleaf shows a generic life cycle model for a product LCA.

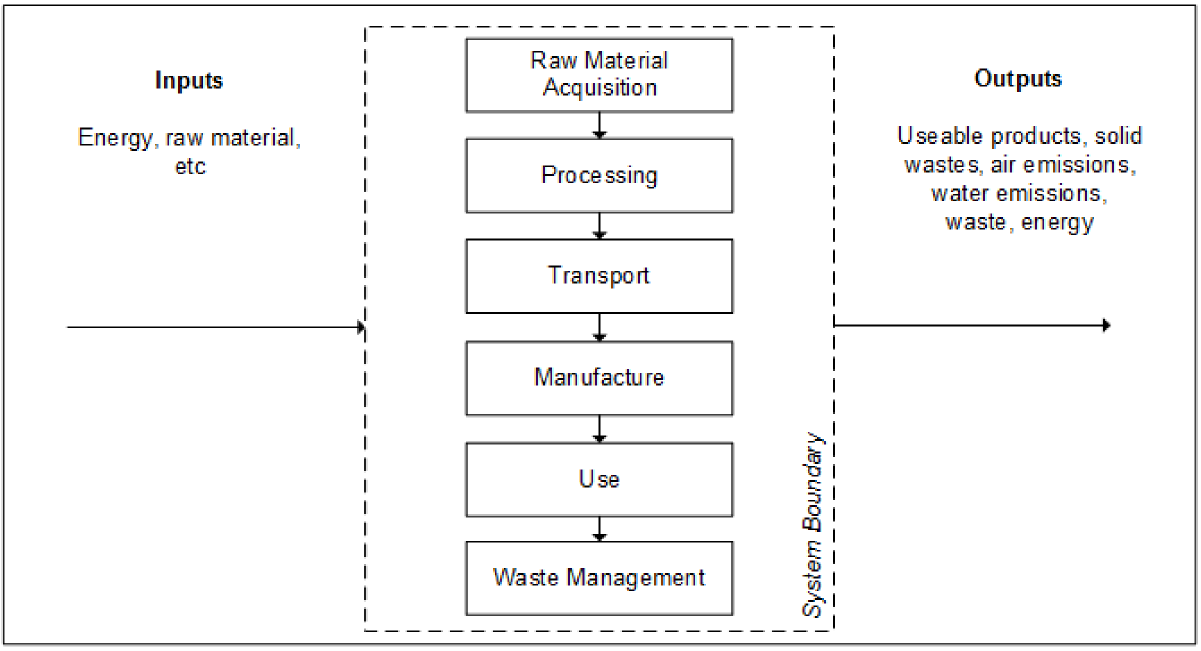

Figure 2.2 Generic life cycle model for a product LCA (Adapted from Baumann & Tillman, 2004:20)

As shown in Figure 2.2, the system boundary encapsulates the collection of operations that define the life cycle of the product or system under study from raw material extraction through to waste management. The system boundary is defined during phase one of the process (the goal and scope definition) and need not necessarily be focused across the entire life cycle. The reasons for conducting a LCA can vary from exploring the life cycle as a whole, allowing the identification of improvement possibilities within the life cycle and understanding which activities contribute the most towards the environmental impact of the product or service, to answering more focused questions regarding individual life cycle stages (Baumann & Tillman, 2004).

2.1.3 Application of LCA to Sustainable Product Design

As previously asserted, the LCM of products has become a regulatory requirement in many countries, informed by the practice of LCA (UNEP & SETAC, 2012). Historically, the direct application of LCA to product design has been constrained by the time consuming nature of the process and demand for specific data of the type typically not available in the early stages of product development (Biswas et al., 1995). Although there has been substantial development in terms of addressing these constraints, particularly in terms of database development, this remains an ongoing process. Such constraints are not, however, unique to LCA, as complexity, time constraint, and lack of necessary environmental knowledge are three typical limitations to the systematic and effective application of various methodologies supporting sustainable design (Bovea & Pérez-Belis, 2012).

While the efficient application of LCA methodology to product design is dependent on a high level of information about the product (Bovea & Pérez-Belis, 2012, Fargnoli & Kimura, 2006), the “early integration of environmental aspects into the product design and development process” has been recognised as a key aspect of sustainable design (Bovea & Pérez-Belis, 2012:68). Given the apparent limitations in the availability of data for the early stages of product development, to address this paradox, various simplified LCA methodologies have been proposed to enable the estimation of environmental impacts early in the design stage of a product’s life cycle (Kaebernick, Kara & Sun, 2003). While there are numerous benefits to a comprehensive LCA compared to a “quick and dirty” screening type approach, when using LCA for design purposes a compromise must typically be made between a “reasonably comprehensive coverage of the life cycle and the time needed for data collection and modelling” (Klöpffer, 2003:158). Although simplified methodologies might lack the depth and/or accuracy of more comprehensive studies, their application is aimed at evaluating the main environmental impacts — or ‘hotspots’ — associated with a product’s life cycle in the initial design stages (Bovea & Pérez-Belis, 2012). The benefit of this application is that it supports the development of various life cycle scenarios, enabling the dynamic evaluation of different concepts and design features (Fargnoli & Kimura, 2006).

While a lack of detailed inventory data presents one challenge to LCA modelling in the context of product design, the geographical dependence of this data presents another. In addition to defining a system boundary for the LCA of a product or process, it is necessary to define a geographical boundary. This geographical definition is necessary, as key contributors to life cycle inventory (LCI) data, such as infrastructure, waste management, and transport systems, tend to vary across different regions (Baumann & Tillman, 2004). In terms of applying LCA as a tool to support sustainable product design, given the increasingly globalised economy, the definition of a geographical boundary therefore requires careful consideration, given that different parts of a product’s life cycle have the potential to occur in different parts of the world (Baumann & Tillman, 2004). Given that the consumer use and end-of-life stages of a product’s life cycle can occur outside of the country where the product is produced (Baumann & Tillman, 2004), in terms of designing a product to have the lowest overall impact, it is necessary that the impacts arising from those stages are assessed relative to the country in which they occur to avoid limiting the applicability of LCA results to a specific context (Friedrich & Trois, 2016).

While it can be assumed that the product itself will be likely to be used more-or-less as intended across different geographies, disposal practices and waste management (the end-of-life stage of a product’s life cycle) can vary between different countries and regions (Hoornweg & Bhada-Tata, 2012). What this implies is that a product designed on the basis of LCA to have the lowest impacts in one country could have unexpected or adverse impacts in another, due to differences in end-of-life management. The accurate assessment of the end-of-life impacts of a product within LCA methodology is therefore considered of critical importance to inform sustainable product design (Leal Filho et al., 2016), particularly as increasing population, wealth and urbanisation drive the consumption of processed goods, thus increasing the generation of waste (Cherubini, Bargigli & Ulgiati, 2009).

2.2 Assessing Product End-of-Life Using a LCA Approach

2.2.1 Overview of the Application of LCA Methodology to Waste Management

While LCA methodology is considered one of the most effective management tools for the identification and assessment of the environmental impacts associated with waste management (Cherubini, Bargigli & Ulgiati, 2009), limited attention has been given to the end-of-life stage of a LCA compared to other stages in a product’s life cycle (Bjarnadóttir et al., 2002). According to Obersteiner et al. (2007), product LCAs commonly exclude end-of-life from the scope of the assessment, or else assume a simplified disposal scenario. However, Obersteiner et al. (2007:S69) go on to criticise this approach, concluding that as waste typically provides a “far from negligible contribution” towards the overall impacts associated with a product’s life cycle, it should by default be included in the evaluation of a product’s life cycle. The increasing quantities of consumer waste being generated globally (Leal Filho et al., 2016) and the constraints in the earth’s capacity to meet the rise in resource demands necessitate the need to design products to promote resource efficiency at the end-of-life stage. Unfortunately, challenges such as data gaps and methodological decisions (Obersteiner et al., 2007) prohibit the inclusion of the end-of-life stage in a product LCA. Although addressing these issues has been the subject of much research, modelling the fate of substances contained in the waste is constrained due to limitations in the current state of scientific knowledge (Obersteiner et al., 2007).

The end-of-life assessment of a product considers the environmental impacts from the point of generation of the waste through to its final disposal. Waste management scenarios for assessing the end-of-life stage in a product LCA can fall within an integrated waste management system, which incorporates different unit processes involved in waste management as well as various waste management strategies. A generic integrated waste management system is shown in Figure 2.3.

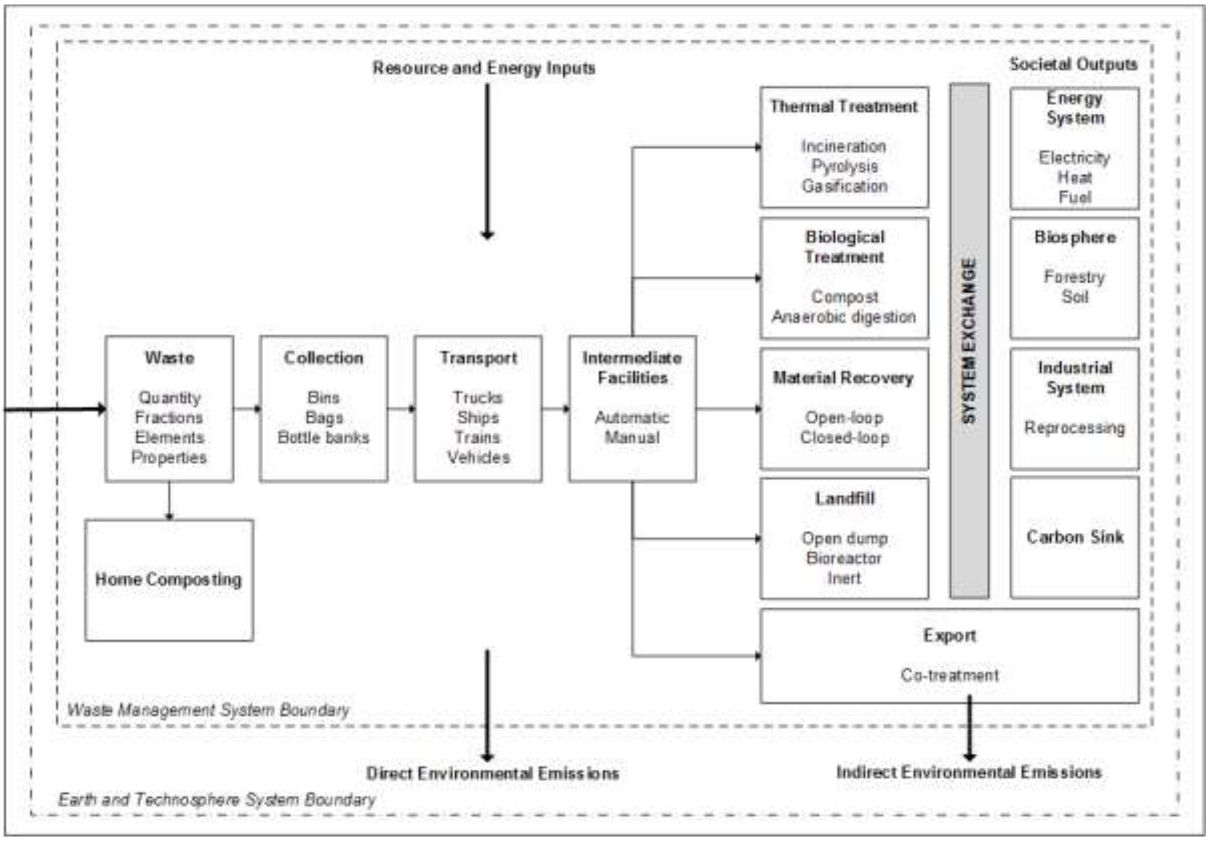

Figure 2.3 Generic waste management system for use in a LCA (Adapted from Gentil et al., 2010:2637)

The generic waste management system defined in Figure 2.3 has application to both product and system LCAs. Where a product LCA models only those impacts incurred from the specific process used to treat/manage the product at its end-of-life, a waste system LCA by contrast assesses the "environmental performance of a number of interconnected waste management technologies based on a specific waste composition from the point of generation of the waste to its final disposal" (Gentil et al., 2010:2637). Both cases, however, use essentially the same LCA methodology, which is applied to both with equal success (Gentil et al., 2010).

From a product design perspective, the results of both product and system LCAs can be of relevance in informing decision-making regarding the end-of-life options for a product. While a product LCA will inform a designer of the impacts associated with a particular treatment option relative to the rest of the product's life cycle, a system LCA illustrates the relative impacts associated with different treatment options for a particular waste or set of wastes. Achieving sustainable waste management therefore requires that directed input be given towards controlling of the flow of waste materials to ensure that resource and energy efficient systems take on an increasingly central role in waste management (Wittmaier, Langer & Sawilla, 2009).

2.2.2 End-of-Life and Waste Management Modelling in LCA

Due to the increasing relevance of LCA as a decision support tool to inform sustainable end-of-life design and the geographic specificities associated with waste management, a review of LCA studies undertaken in this field provides various insights regarding the application of this methodology. Not only does it provides insight into the geographic distribution of such studies, but further illustrates how different waste management systems can be modelled and the results thereof (Laurent et al., 2014a). According to Laurent et al. (2014a:574), although a number of review studies have been undertaken, these do not provide a comprehensive coverage of the field as a whole, as their focus tends to be "limited to specific methodological aspects or specific types of waste or specific waste management systems." An overview of review studies, as presented by Laurent et al. (2014a) is shown in Table 2.2.

Table 2.2 Overview of existing review studies for the application of LCA to the field of waste management (Adapted from Laurent et al., 2014a:574)

Study	Focus of Study	Number of Studies/Models Reviewed
Villanueva and Wenzel (2007)	Review of LCA studies assessing the management of paper and cardboard waste	9
Cleary (2009)	Review and analysis of methodological conduct and findings of LCA studies assessing the management of municipal waste	20
Lazarevic et al. (2010)	Analysis of LCA studies assessing the management of post-consumer plastic waste in Europe	10
Michaud et al. (2010)	Review of findings of LCA studies focusing on the benefits of recycling	55
Gentil et al. (2010)	Overview of existing models for LCA applied to solid waste	8
Bernstad and la Cour Jansen (2012)	Review of 25 LCA studies of bio-waste treatments	25
Morris, Jeffrey, Matthews and Morawski (2013)	Meta-analysis of 82 studies assessing the management of organic waste	82

Although the number of studies reviewed under each specific focus outlined in Table 2.2 illustrate the uptake and application of LCA within the field of waste management, major conclusions drawn from these reviews confirm that issues relating to waste management and recycling are a recurring theme across LCAs (Laurent et al., 2014a). System boundary definition for waste treatment processes has been recognised as a key challenge in this regard, particularly in terms of the allocation of burdens between different life cycles (Laurent et al., 2014b). Additional challenges include time perspectives, impact modelling, and the lack of reliable input data (Ekvall et al., 2007).

With regards to waste management, while the international standardisation of LCA has come some way in reducing the perceived arbitrariness of the methodology (in particular with regards to managing time frames and impact modelling), important methodological choices remain to be made in individual studies (Ekvall et al., 2007). These choices have the potential to influence the results of the study and as such, because these methodological choices can be influenced by the LCA practitioner or commissioner, the findings are not necessarily objective (Ekvall et al., 2007).

To address the shortcomings of previous review studies in capturing the overall state of the field of LCA applied to solid waste management, an exhaustive critical review of 222 LCA studies of solid waste management systems reported in scientific literature and public reports was undertaken by Laurent et al. (2014a). A significant finding arising from this review is the notable imbalance in the geographical distribution of the reviewed studies, with the majority concentrated in Europe with little representation of waste management systems in developing countries (Laurent et al., 2014a). No waste treatment LCAs were identified for Africa, their absence suggesting a poor penetration of LCA into these regions (Laurent et al., 2014a). However, these findings regarding the availability of waste-focused LCAs within Africa are not entirely accurate. For example, within South Africa some such studies exist (i.e. Friedrich & Trois (2013a), Friedrich & Trois (2013b), Friedrich & Trois (2016), von Blottnitz & Petrie (1995) and Rwodzi & von Blottnitz (2000)). Thus, although the availability of African focused waste LCA studies should be acknowledged, their relative paucity is not disputed.

It has been suggested that the limited penetration of LCA into developing countries can in part be explained by the lack of directives (or lack of uptake of existing directives) aimed at including LCT into waste management strategies in comparison to those that exist in Europe (Laurent et al., 2014a). Furthermore, data from developing countries is limited (Karak, Bhagat & Bhattacharyya, 2012). As LCA data best represents conditions in industrialised countries, LCA studies making use of this data are likely to be biased towards these conditions (Hertwich, 2005). Given the variation in waste management structures and practices that occurs between developed and developing countries, LCA studies in developed countries are unlikely to be applicable to situations in developing countries (Laurent et al., 2014a). Therefore, although limitations in data availability is strongly limiting with regards to the uptake of LCA methodology, the current geographical imbalance in LCAs highlights the need for more LCA waste studies to be undertaken in developing countries (Laurent et al., 2014a).

It was further asserted that across the 222 reviewed LCA waste studies — with the exception of landfilling, which consistently represents the least environmentally favourable management option — there exists no definitive agreement with regards to which waste treatment technology performs better for the four major waste types considered (namely plastic, organics, paper and mixed streams) (Laurent et al., 2014a). This observation indicates the dependence of the context and local specificities on the LCA results, thus limiting the generalisation thereof (Laurent et al., 2014a). The generalisation of LCA results can be attractive for such purposes as developing appropriate management strategies and designing products for sustainability in the end-of-life stage without requiring decision-makers to undertake intensive LCAs (Winkler & Bilitewski, 2007). However, more accurate results would be achieved with the adaptation of LCA studies to represent local conditions and factors. To avoid limiting the applicability of LCA results to a specific context, it has been suggested that the relevant LCA model be adapted to take into account, amongst else, site-specific waste composition, treatment efficiencies, and the local energy mix (Laurent et al., 2014a).

2.2.3 Tools for End-of-Life and Waste Management Modelling

Given the popularity of LCA for application to waste management modelling, since the early 1990s, a variety of dedicated LCA waste models and software tools have been developed (Gentil et al., 2010, Kulczycka et al., 2015, Winkler & Bilitewski, 2007). The benefits associated with the use of dedicated waste models are largely directed towards facilitating the accessibility of LCA in this application. Not only does the use of a dedicated model speed up the analysis, but further promotes the use thereof by allowing decision makers and waste managers to use LCA without a detailed knowledge of the methodology as a pre-requisite (Winkler & Bilitewski, 2007).

LCA waste models vary in terms of “applicability, functionality, licensing restrictions and costs” (Gentil et al., 2010:2637). From a generic perspective, it is expected that the model link the specific elements contained in a waste to the emission output from the relevant waste handling, treatment, or disposal process (Gentil et al., 2010). To achieve this function, it is necessary that the model to possess the following capabilities (as according to Gentil et al. (2010:2637)):

- Models must respond to changes in waste composition and represent elemental waste specific emissions in addition to waste processes' specific operating emissions.
- Models should account for emission offsets with other systems by including substitution with energy systems and manufacturing of primary resources.
- Models should display flexible system boundaries, include a country-specific energy mix and include the assessment of an integrated and interconnected system (i.e. the inventory for all transportation and waste management processes from collection to final disposal).

Despite these generic requirements, dedicated waste models typically "suffer a lack of harmonisation" (Gentil et al., 2010:2636). This has been attributed in part to the various complexities of waste modelling and the range of country-specific data required to represent local practice, leading to these models being developed in relative isolation (Gentil et al., 2010). The impact this has on the results obtained from the use of different models was first highlighted by Winkler and Bilitewski (2007), who undertook an investigation into six dedicated LCA waste models with the objective of illustrating that different LCA tools do not lead to "different or contradictory conclusions and that variation in results [of the LCAs] are within an acceptable range" (Winkler & Bilitewski, 2007:1022). However, the results of this study — in which the waste management system of Dresden in Germany was assessed — found instead high variability in the results, leading to contradictory conclusions between models. This variability was attributed in part to the limitations within the models' ability to represent the waste management system in its entirety and all environmentally relevant aspects thereof (Winkler & Bilitewski, 2007). The apparent discrepancy was exacerbated by the complexity of waste management systems, requiring a number of assumptions to be made, resulting in the oversimplification of the system and generation of results not reflective of reality.

Based on the result of the study undertaken by Winkler and Bilitewski (2007), it was asserted by Gentil et al. (2010) that if the assumptions made and possible calculation errors are capable of generating such large discrepancies, it is necessary to identify key criteria with the potential to impact on the results of waste models. To address this need, Gentil et al. (2010) undertook a review of nine waste LCA models, with an emphasis on the methodology used, input parameters, and modelling assumptions in order to identify those key criteria affecting the model output. The findings from this review highlighted the influence of several criteria including the functional unit, system boundaries, waste composition, energy modelling, and modelling assumptions applied to the representation of the waste management system as having a significant influence on the model output.

While some of the differences observed by Gentil et al. (2010) in the reviewed models were attributed to the temporal distribution of their development (with more recent models reflecting advances in the field and benefitting from past work of a similar nature), the geographical specificities of the models was further isolated as a contributory factor. With regards to the latter, because models are optimised for a specific geography (typically their country of development), the output is best representative of processes occurring within the same geographical context (Gentil et al., 2010). From this, a caveat regarding the use of LCA models in an alternative geographical context to which they were developed is issued: the use of country-specific data and assumptions applied in the model development will lead to differences between the results obtained using different models (Gentil et al., 2010).

While dedicated LCA waste models offer one approach to the application of LCA to waste management and end-of-life modelling, it is also possible to use generic LCA software of the type typically applied to a product LCA such as SimaPro, GaBi, Team and Umberto (Kulczycka et al., 2015). While the use of dedicated waste models is typically applied to waste system LCAs, given that a product LCA requires representation of the entire life cycle as opposed to a single stage, the use of generic LCA software can be applied to both models (Kulczycka et al., 2015). However, generic LCA software is not exempt from the challenges facing waste modelling. Both product end-of-life modelling and waste system modelling using generic LCA software is dependent on a detailed LCI. Collecting the relevant data for waste management processes can be both “time consuming and labour intensive” (Gentil et al., 2010:2641). As such, generic LCA software usually contains comprehensive databases that can be used to provide default datasets for various processes (Kulczycka et al., 2015). The representation of certain processes within these databases can be limited and, furthermore, incapable of representing regional specificities (Kulczycka et al., 2015).

2.3 Mapping Municipal Waste Management

2.3.1 Distribution of Waste Management Practices

Global disparities in wealth, and economic and industrial development, amongst other factors, influence waste management practices employed in different regions (Hoornweg & Bhada-Tata, 2012). While economic considerations, such as capital and operating costs, represent a key factor in determining preferential waste management practices, additional factors include the waste composition, land availability, and environmental regulations of the country or region (Doka, 2003c). A comparison of municipal waste management by country income level is shown in Figure 2.4.

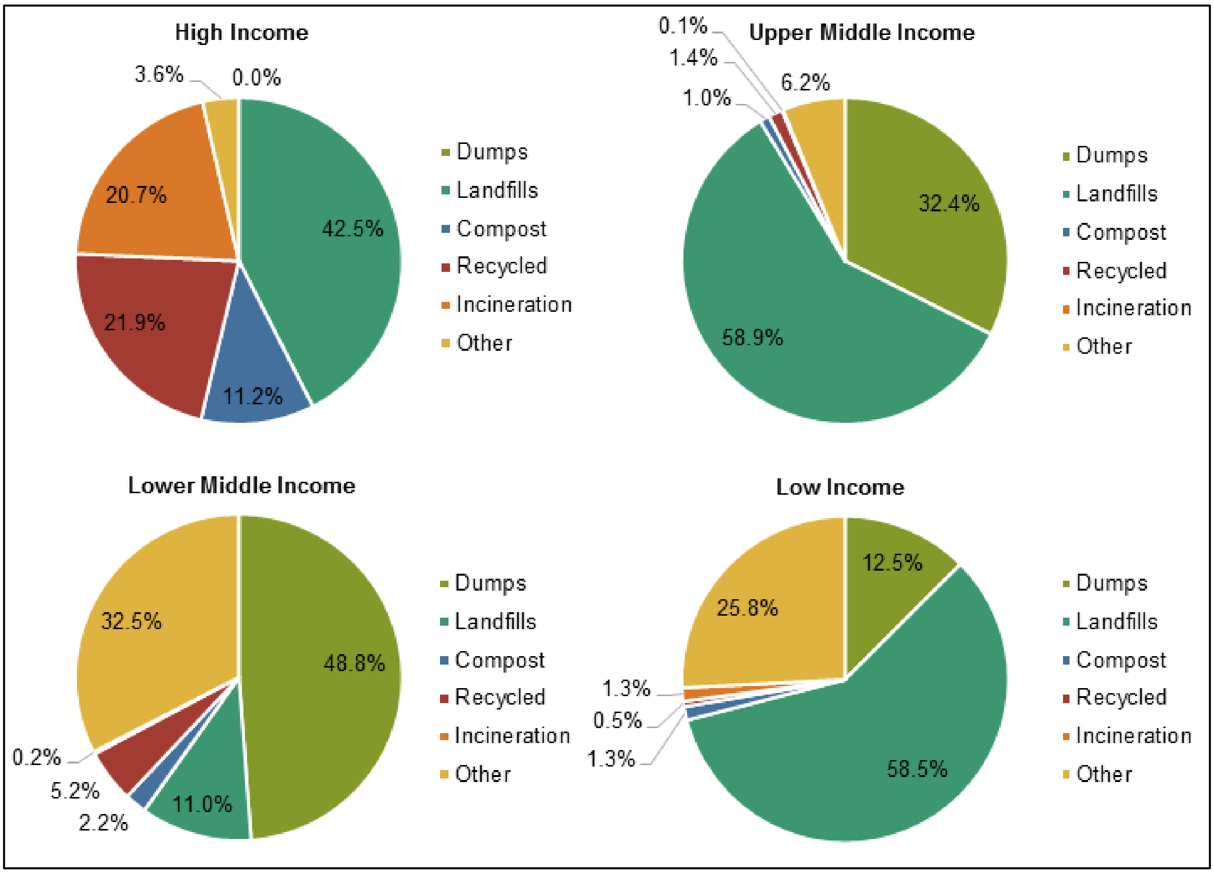

Figure 2.4 Municipal solid waste disposal practices by country income level based on 2010 data (Adapted from Hoornweg & Bhada-Tata, 2012:23)

From Figure 2.4 it can be observed that across each economic classification, landfill disposal represents the most utilised management option for MSW, with the exception of lower middle income countries (reportedly skewed by the inclusion of China's disposal data, which reflects a strong dependence on dumping (Hoornweg & Bhada-Tata, 2012)). While disposal to land (landfill and dumping) is the most utilised disposal option across all four economic classifications, high income countries reflect the greatest proportion of waste directed towards alternative management options. The uptake of alternative waste management practices in high income countries is partially reflective of increasingly stringent regulatory and/or environmental requirements (Leal Filho et al., 2016) and is typically aligned with the principles of the waste hierarchy, intended to improve the sustainability of waste management, as highlighted in documents such as the *Global Waste Management Outlook* (UNEP, 2015). While the shift towards more environmentally sustainable waste management practices is desirable, the elevated infrastructure and operating costs coupled to stringent operating requirements typically associated with alternative management practises are prohibitive to their uptake in developing countries (Leal Filho et al., 2016). A compendium of the relative cost requirements and environmental impacts of the main waste management options utilised in different countries is shown in Table 2.3.

Table 2.3 Compendium of regional distinctions in municipal waste management practices (As presented in Leal Filho et al., 2016:4379))

Region	Main Waste Management Technologies/Practices	Environmental Impact	Cost Level
Developing countries (Africa, Asia, Latin America)	Open dumping Limited incineration (e.g. China) Evolving towards engineered sanitary landfilling	Very High Medium High	None High Low
Eastern Europe (Russia, Ukraine, Belarus)	Sanitary landfilling Limited recycling	High Medium/Low	Medium Medium
European Union	Evolving towards limited landfilling Material and biological recycling Mechanical-biological treatment/incineration Increasing anaerobic digestion	High Low Medium Low	Low Medium High High
North America, Australia	Sanitary landfilling Material recycling Low rates of incineration, mechanical-biological treatment, and anaerobic digestion	High Low Medium	Medium Medium High
Japan	Incineration Material recycling Landfilling of residuals	Medium Low High	High Medium Medium

Table 2.3 not only illustrates the discrepancy in waste management between different countries, but further shows the relative environmental benefits associated with alternative waste management practices to landfill. However, the cost differential between landfill and alterative practices limits their applicability in developing regions. Comparison of Figure 2.4 with Table 2.3 shows that despite the high environmental impacts associated with landfill disposal, it remains a prominent disposal option across the world. Even where sanitary landfill practices are in place, the environmental impacts are still relatively high. These findings are echoed in those presented by Laurent et al. (2014a), which show on the basis of LCA, that landfilling consistently represents the least environmentally sustainable management option for the majority of general waste types. Given the differences in the environmental performance of waste management options, it therefore follows that a product could have different environmental implications depending where it is disposed of.

An additional source of data useful for mapping municipal waste management are the regional defaults reported by the IPCC (2006a) for the relative fraction of MSW managed/treated under various waste management options. These are shown in Table 2.4. Whilst regionalised waste disposal data is useful in mapping MSW disposal in a particular region, analysis of Table 2.4 suggests that there are limitations to the accuracy that can be achieved — for the majority of regions, a relatively large proportion of waste is reported as “unspecified”. Furthermore, for certain regions, notably Africa, limited regional data is available, resulting in limited regional disaggregation.

Table 2.4 Regional defaults for the proportion of municipal solid waste managed by different waste management options (Adapted from IPCC, 2006a:2.5)

Region	Solid Waste Disposal	Incineration	Composting	Unspecified[c]
	Percentage of Solid Waste per Management Option			
Eastern Asia	55	26	1	18
South-Central Asia	74	-	5	21
South East Asia	59	9	5	27
Africa[a]	69	-	-	31
Eastern Europe	90	4	1	2
Northern Europe	47	24	8	20
Southern Europe	85	5	5	5
Western Europe	47	22	15	15
Caribbean	83	2	-	15
Central America	50			50
South America	54	1	0.3	46
North America	58	6	6	29
Oceania[b]	85	-	-	15

[a] A regional average was given for the whole of Africa as data was reportedly not available for more detailed regions within Africa
[b] Data for Oceania are based only on data from Australia and New Zealand
[c] Includes data on recycling for some countries

As both Table 2.3 and Figure 2.4 explicitly report MSW management practices, it is unlikely that informal waste management practices are considered. While Table 2.4 includes a category for unspecified treatment, it is uncertain whether this category encompasses informal waste management practices or not. MSW collection efficiencies can vary significantly between different countries. For example, low-income countries purportedly collect approximately 43% of MSW (with certain African and South American countries reporting collection efficiencies less than 10%), while upper-income countries report collection rates upward of 85% (Hoornweg & Bhada-Tata, 2012:46). This implies that particularly in developing countries, a potentially large fraction of MSW is disposed of informally, either into unregulated open dumps, or else managed by uncontrolled or unregulated treatment such as open burning. Consideration of the informal proportion of waste and the impacts associated with its disposal could therefore be significant when assessing the impacts associated with product disposal.

2.3.2 Municipal Waste Composition

In addition to supporting differences in waste management practices, the income and urbanisation of a country or region is also recognised as having an influence on the composition of the MSW stream. An illustration of the variation in MSW composition with country income level is shown in Figure 2.5 overleaf.

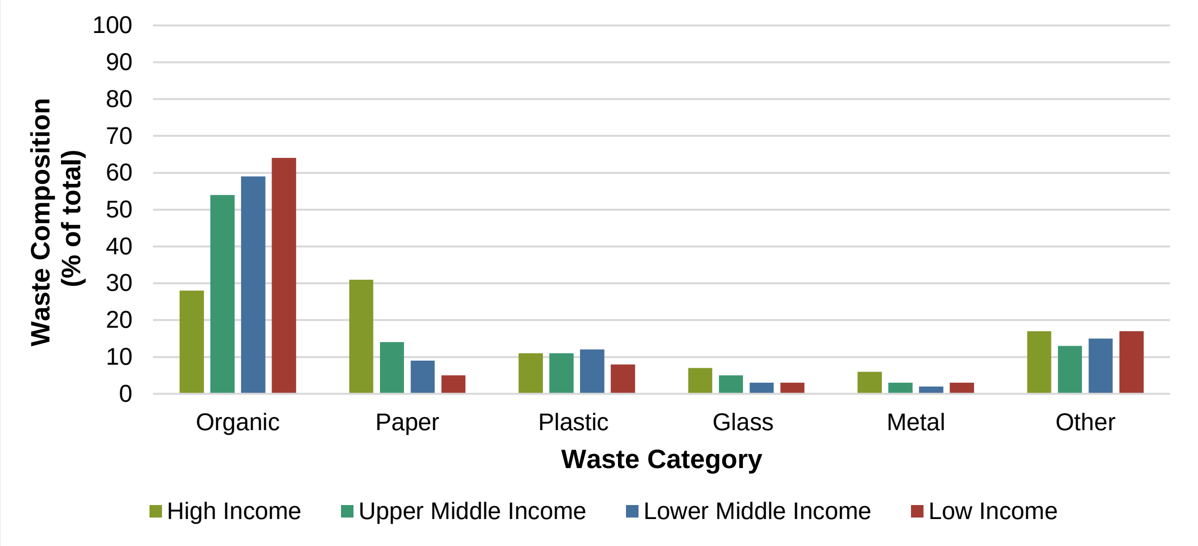

Figure 2.5 Waste composition based on country income for 2010 (Adapted from Hoornweg & Bhada-Tata 2012:19)

According to Figure 2.5, the relative proportion of organic waste in MSW shows the most notable variation, increasing with decreasing income level. It is generally accepted that a growth in gross domestic product (GDP) and urbanisation results in an increase in the relative proportion of inorganic waste in MSW, as increased consumer wealth and buying power drives consumption of packaged consumer goods and processed food (Cohen, 2017, Hoornweg & Bhada-Tata, 2012). This trend is supported by the regionalised waste composition data presented by the IPCC (2006a:2.12), which suggests that developing regions such as Middle Africa and South East Asia have notably higher organic (in this case, specifically food waste) fractions (43.4% and 40.3% respectively) than developed regions such as Northern Europe and North America (23.8% and 33.9% respectively). In addition to economic development, the composition of MSW can be further influenced by factors including cultural norms, geographical location, energy sources and climate (Hoornweg & Bhada-Tata, 2012). Not only does the composition of MSW affect the applicability of waste management options but can further influence the environmental performance thereof (Bhailall, 2015, Doka, 2003d, Hoornweg & Bhada-Tata, 2012). With regards to the latter, the effect of waste composition on the polluting potential of landfill sites in particular can be significant, with increased organic content enhancing the waste decomposition process and increased inorganic content resulting in increased inorganic pollutant loads in the landfill leachate (Shamrock, 1998).

2.3.3 Municipal Waste Generation Rates

While the discussion in Sections 2.3.1 and 2.3.2 enables a general mapping of municipal waste management and waste flows, waste generation rates (WGRs) are beneficial in quantifying the amount of waste generated and treated in a country or region. Such an approach is particularly beneficial in the absence of reliable municipal waste reporting or country specific waste data. WGRs are linked to a number of factors including population size, economic productivity, and the population distribution in terms of urban and rural settlement (Bogner et al., 2008). Economic factors in particular are recognised for their influence on WGRs (Cointreau, 2006, Hoornweg & Bhada-Tata, 2012), with regional variations in WGRs reflective of the relative wealth of a particular region (Hoornweg & Bhada-Tata, 2012).

Given the potential benefit of WGRs in the quantification of waste flows (thus supporting waste management policy and decision-making), various models and approaches are available to model MSW generation. However, a comprehensive review of 45 published models for MSW generation undertaken by Beigl, Lebersorger and Salhofer (2008) suggests that existing models are highly heterogeneous, using various economic, socio-demographic or management orientated data with the modelling approach ranging in complexity from linear relationships to multivariate modelling. Application of these models requires the availability of the requisite data, which is typically lacking in developing countries (Hoornweg & Bhada-Tata, 2012), hence limiting the use of such models in these regions.

In lieu of suitable data to enable the use of a predictive model, regional per capita MSW generation rates have been proposed by both the IPCC (2006a), and Hoornweg and Bhada-Tata (2012) (who further present income-specific WGRs). Both sources acknowledge the lack of available and accurate waste data in developing countries — particularly in Africa — as a limitation in the determination of regional WGRs. A comparison of the WGRs presented by these sources is shown in Table 2.5. However, differences in the regionalisation utilised by the different sources limits the direct comparison that can be undertaken between the reported MSW generation rates.

Table 2.5 Municipal solid waste generation rates for different regions (Adapted from IPCC, 2006b:2.5 and Hoornweg & Bhada-Tata, 2012:9)

Source	Region	Municipal Solid Waste Generation kg/capita/day
IPCC (2006a)	Eastern Asia South-Central Asia South East Asia	1.01 0.58 0.74
	Africa[a]	0.79
	Eastern Europe Northern Europe Southern Europe Western Europe	1.04 1.75 1.42 1.53
	Caribbean	1.34
	Central America South America North America	0.58 0.71 1.78
	Oceania[b]	1.89
Hoornweg and Bhada-Tata (2012)	Africa[c]	0.65
	East Asia and Pacific	0.95
	Eastern and Central Asia	1.1
	Latin America and the Caribbean	1.1
	Middle East and North Africa	1.1
	Organisation for Economic Co-Operation and Development (OECD)	2.2
	South Asia	0.45
	Income-Specific Waste Generation Rates	**Municipal Solid Waste Generation kg/capita/day**
	High	2.1
	Upper Middle	1.2
	Lower Middle	0.79
	Lower	0.60

[a] A regional average was given for the whole of Africa as data was reportedly not available for more detailed regions within Africa
[b] Data for Oceania are based only on data from Australia and New Zealand
[c] Based on Sub-Saharan regional data

Table 2.5 suggests that there are notable differences in WGRs between regions, with MSW generation rates consistently higher in developed regions than developing regions. Due to the paucity of waste data in Africa, for both sources, a regional average for the continent as a whole was reported in place of more detailed regional estimates. The reported lack of waste data goes some way in explaining the discrepancy between the WGRs reported for Africa. Hoornweg and Bhada-Tata (2012:8) explicitly note that their African WGR (0.65 kg/capita/day) is based on Sub-Saharan data. The IPCC (2006a) estimate of 0.79 kg/capita/day does not explicitly mention which region it represents.

2.4 Representing Landfill Disposal with LCA Methodology

2.4.1 Overview of Conventional Landfill Practices

As discussed in Section 2.3, due largely to the comparatively high costs associated with alternative disposal globally, landfilling remains dominant in terms of MSW disposal. Historical dependence on landfill operations coupled to the increasing demand for improved environmental management has resulted in the operation of a number of different types of landfill sites (Obersteiner et al., 2007). Conventionally, landfill sites can be classified as either sanitary, unsanitary, or open dumps. While the impacts associated with conventional landfill processes are always relatively high (Laurent et al., 2014a), the emission output from a landfill (and hence impact potential) is strongly dependent on the physical infrastructure and operating procedures of the site (Manfredi & Christensen (2009), Kirkeby et al. (2007), Friedrich & Trois (2013b)). A comparison of the key differences with regards to the physical-infrastructure and operation of different types of conventional landfill sites is shown in Table 2.6.

Table 2.6 Comparison of the characteristics of different conventional landfill types for the disposal of municipal solid waste (Adapted from Doka, 2016:16 and Frøiland Jensen & Pippatti, 2001:425)

Infrastructure/Operational Process	**Landfill Site**		
	Sanitary Landfill	**Unsanitary Landfill**	**Open dump**
Excavation pit	Yes — minimum depth of 10 m	Yes — variable depth; shallow sites <5 m	No — variable depth; no formal excavation
Bottom liner	Yes	No	No
Waste compaction	Yes — high compaction with suitable equipment	Yes — poor compaction using light operational equipment	No
Surface covering	Yes — frequent and adequate cover	Yes — limited	No
Scavenging at operational area (human and animal)	No	Yes	Yes
Prevention of landfill fires	Yes	Limited	No — frequent fires often deliberate and systematic to reduce waste volumes
Litter Control	Yes	Limited	No
Leachate collection and treatment system	Yes — well designed and well operated	No	No
Storm water systems	Yes	No	No
Landfill gas collection tubes	Yes	No	No
Landfill gas capture and utilisation	Yes — varying level of utilisation	No	No

According to Table 2.6, sanitary landfill sites are typically well-managed with various barrier and containment systems in place to manage the emissions from the site, and operational controls in place to prevent the occurrence of scavenging and fires. By contrast, unsanitary landfill sites and open dumps reflect reduced infrastructure, a lack of barrier and containment systems, and limited operational controls. It can therefore be inferred that the characteristics of different landfill sites will influence both how emissions are released from the landfill (for example, whether as direct air or water emissions) and the quantity of specific emissions. For the sanitary landfill, both landfill gas (LFG) and leachate emissions are typically captured for further treatment, whereas for unsanitary practices and open dumps, these emissions are released directly into the air or the sub-soil beneath the waste body.

2.4.2 Assessing Landfill Performance with LCA

According to Manfredi and Christensen (2009), for conventional landfill operation[1], direct emissions of leachate and LFG from the waste body are the highest contributors towards the environmental impacts of the site. These emissions predominantly result in the pollution of groundwater resources in the case of the former, with the latter contributing towards greenhouse gas (GHG) emissions, fire hazards, odours, and vegetation damage (Manfredi & Christensen, 2009). While a proportion of these emissions occur within a relatively short time frame after disposal, landfill sites represent a potential source of significant long-term emissions (Laner, Fellner & Brunner, 2011). Given that emissions can continue to occur for centuries beyond waste deposition, the environmental consequences of landfilling can be postponed to future generations (Doka & Hischier, 2005). The implications of the emission potential of landfill sites are discussed in Section 2.4.3. A comparison of the different sources of impacts associated with conventional landfill technology, for a 100 year time horizon, is shown in Figure 2.6.

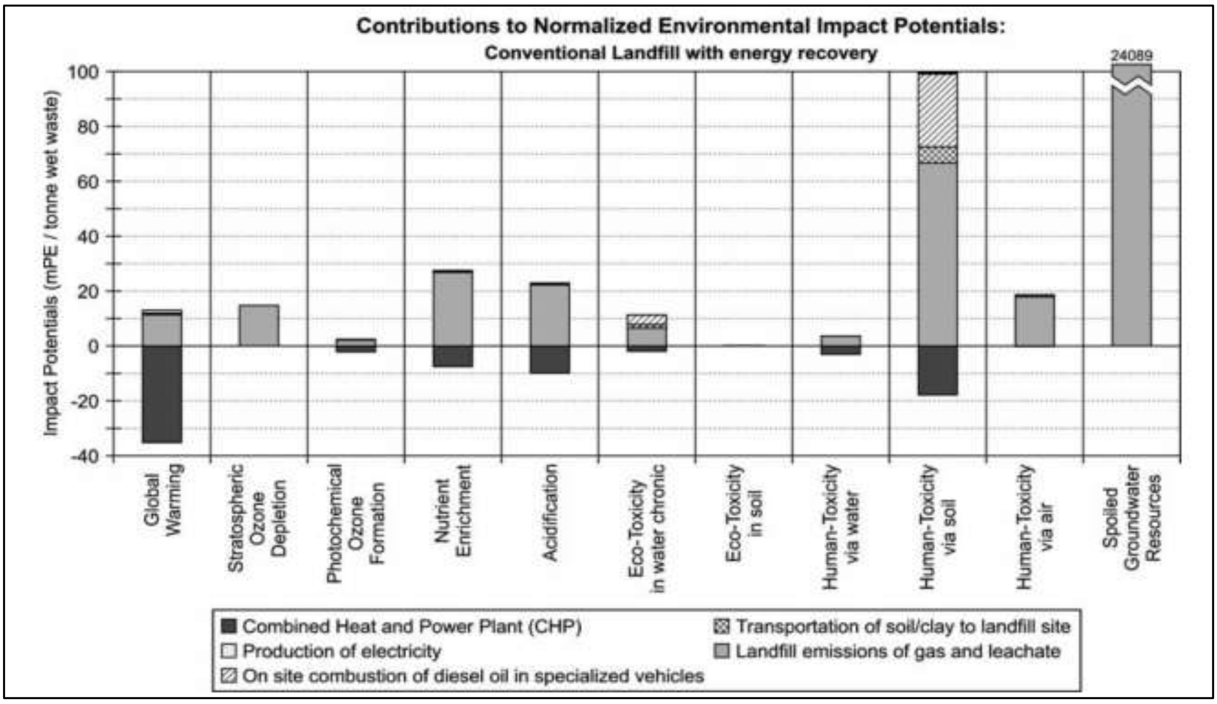

Figure 2.6 LCIA of a conventional landfill with energy recovery as normalised impact potentials (for standard and toxicity related environmental impact categories) for the time horizon 0 – 100 years (As presented in Manfredi & Christensen, 2009:38)

[1] Conventional landfill operation — as defined by Manfredi and Christensen (2009) — does little to affect the waste degradation and subsequent leachate and gas emissions, but rather implements technical measures to contain and treat these emissions. Technical measures in this sense include bottom liner, leachate collection system, leachate treatment prior to discharge, top soil cover, gas collection system, flares, and gas utilization for energy recovery.

As evidenced in Figure 2.6, process-linked burdens arising from site infrastructure and daily operations on the site — such as the combustion of diesel oil in vehicles operating on the site and the transportation of clay/soil cover material — have a relatively low impact potential compared to those arising from the waste itself. In particular, the relative contribution of LFG emissions towards the overall impacts of the site is supported by the results of a similar investigation undertaken by Kirkeby et al. (2007).

Given the relative importance of leachate and LFG emissions towards the overall environmental impact from a landfill site, a sensitivity analysis was undertaken by Kirkeby et al. (2007), in which the effect of various parameters on the emissions and subsequent environmental impact from a reference landfill scenario[2] was investigated. The results of this investigation (Figure 2.7) suggest that the overall environmental impact of landfilling is most sensitive to the LFG collection efficiency, and use thereof (Kirkeby et al., 2007). Somewhat unexpected is the observation that the amount of leachate generated has limited effect on the impact potential of the site. However, comparison of the impact categories considered by Kirkeby et al. (2007) (Figure 2.7) to those presented by Manfredi and Christensen (2009) (Figure 2.6) suggest that the impact of leachate is seen predominantly in spoiled groundwater resources. Thus the exclusion of this category by Kirkeby et al. (2007) might underestimate the effect that increased leachate generation has on the environmental impact of landfilling.

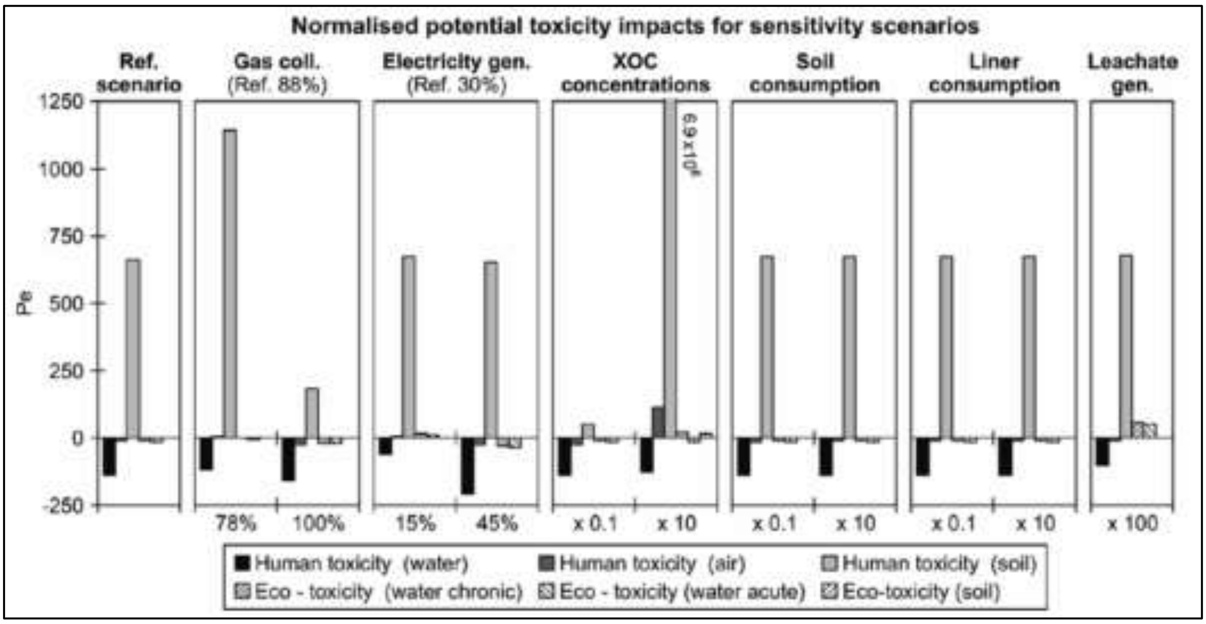

Figure 2.7 Normalised environmental impacts for sensitivity scenarios to a reference conventional sanitary landfill scenario (As presented in Kirkeby et al., 2007:969)

The inability of traditional landfill practices to meet sustainability targets has seen increased implementation of engineered systems to improved control over the emission output (Manfredi & Christensen, 2009). Different technologies reflect different approaches with regards to overall impact reduction. Conventional engineered controls typically have limited impact on the waste degradation that occurs, but instead implement various technical measures to manage the subsequent emissions (Manfredi & Christensen, 2009). Technical measures include the use of a bottom liner, top soil cover, and collection and treatment systems for both LFG and leachate (Manfredi et al., 2009a).

[2] Here the reference scenario – as defined by Kirkeby et al. (2007) – assumes LFG collection efficiency of 88%, electricity generation efficiency of 30%, transportation for soil consumption of 0.50 tonne/tonne, liner consumption of 0.024 kg/ton and a leachate generation rate of 0.5 m/year for years 11 – 100.

Alternative landfill technologies by contrast include bioreactor and semi-aerobic technologies, in which the waste degradation process is enhanced to achieve a “faster and more extensive stabilization of waste” within a reduced time frame, as opposed to conventional landfilling (Manfredi & Christensen, 2009:33). The effect of alternative landfill technologies on the environmental performance of different landfill sites was investigated by means of LCA by Manfredi and Christensen (2009). The results of this investigation for standard and toxicity related impact categories are shown in Figure 2.8, with the impact potential on spoiled groundwater resources shown in Figure 2.9.

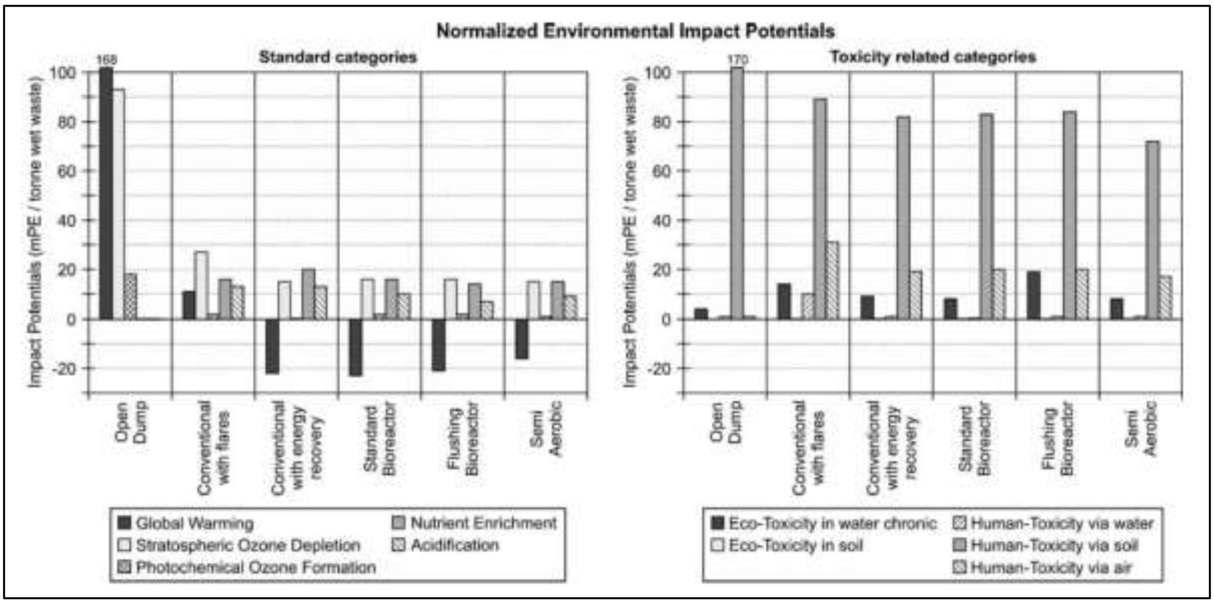

Figure 2.8 LCIAs of six landfilling technologies as normalised impact potentials (standard and toxicity related environmental impact categories) for the time horizon 0 – 100 years (As presented in Manfredi & Christensen, 2009:39)

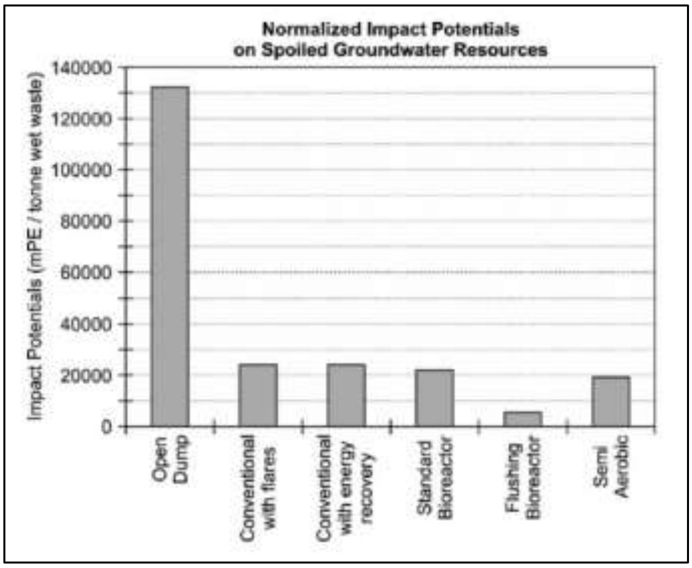

Figure 2.9 LCIAs of six landfilling technologies as normalised impact potentials (impact on spoiled groundwater resources) for the time horizon 0 – 100 years (As presented in Manfredi & Christensen, 2009:39)

Both Figure 2.8 and Figure 2.9 illustrate how the environmental impacts of a landfill site can be reduced with various controls and management measures. According to Figure 2.8, conventional or alternative technologies significantly reduce the environmental impacts in standard and toxicity related impact categories when compared to open dumping (included as a worst case scenario). The recovery and utilisation of LFG in particular is beneficial, illustrating the potential to result in saved emissions and avoided impact potential (as shown by the negative values in the graphs) (Manfredi & Christensen, 2009). In terms of leachate management, Figure 2.9 indicates that the collection of leachate to avoid infiltration into groundwater has important implications in reducing the impact of the site on spoiled groundwater resources.

The results presented by Manfredi and Christensen (2009) are supported by independent studies by Cherubini, Bargigli and Ulgiati (2009), and Kirkeby et al. (2007). These studies also compare the environmental performance of different landfill operations and technologies for the treatment of MSW using LCA methodology, with the results illustrating both a reduction in environmental impacts with improved emission management and the apparent benefit in the recovery and utilisation of LFG.

In addition to the type of technology implemented on a landfill site, landfill emissions are directly dependent on the type of waste that is deposited. It has been suggested that CH_4 produced during the degradation of organic matter is the largest contributor towards GHG emissions from the waste sector (Bogner et al., 2008). Indeed, various policies and strategies, particularly within the European Union, have been directed towards reducing the quantity of biodegradable waste that is landfilled in an attempt to reduce emissions from this sector (Bogner et al., 2008). The potential benefit of “low-organic-carbon waste landfills” has been illustrated by Manfredi et al. (2009a). In addition to GHG emissions consequent of the degradation of organic matter, the leaching of toxic materials (both organic and inorganic) from landfilled waste has been recognised for its significant contribution towards surface and groundwater contamination (Scott et al., 2005). Inert waste fractions by contrast do not have any noticeable impact on the environment (Manfredi & Christensen, 2009).

2.4.3 Assessing Long-Term Landfill Emissions in LCA

Unlike many other processes that are represented in a LCA, emissions from a landfill have the potential to continue for centuries or millennia beyond its operational phase (Doka & Hischier, 2005). Figure 2.6 – Figure 2.9 all illustrate the relative impacts associated with landfill sites when considering a time horizon of 100 years. However, consideration of a longer time horizon is likely to influence these results (Manfredi & Christensen, 2009). This assertion is based on the results of mass balance calculations showing that a large proportion of substances in household waste remain stored in the waste body at the end of the 100 year time horizon[3] (Manfredi & Christensen, 2009). This observation is supported by both Hellweg (2000) and Doka and Hischier (2005). Following this observation, it has been suggested that the consideration of a longer time horizon — for example 500 years — is likely to increase the fraction of organic matter that is emitted in LFG or leachate, with a longer time horizon still, increasing the release of heavy metals (Manfredi & Christensen, 2009).

Given the potential for landfill emissions to occur for an extensive time period beyond the deposition of waste, the clear definition of a time frame is critical when modelling a landfill process in a LCA (Obersteiner et al., 2007). Having defined a time frame, the impact potential of the waste can be determined by integrating the potential emissions over this time period (Finnveden, 1999). While general consensus has been reached that emissions should be integrated over a foreseeable period, the time frame utilised in LCA studies has been found to vary from as little as 15 to up to 100 000 years (Obersteiner et al., 2007:S60). The selection of a time frame can pose an ethical debate, as limiting the time frame considered lessens the emissions from the site (Obersteiner et al., 2007).

[3] For example, approximately 50% of carbon and 99% of heavy metals remain stored, as according to Manfredi and Christensen (2009:42).

The definition of a longer time horizon is challenging due to the absence of any empirical data for long-term emissions (Obersteiner et al., 2007). Both the availability and reliability of data decrease drastically with increasing landfill age, thus compromising the quality of the LCA results (Manfredi & Christensen, 2009). The CH_4 generating anaerobic phase of waste decomposition (giving rise to CO_2 and CH_4; both major constituents of LFG) typically occurs for 30 years after waste deposition (Doka, 2003d). From a future emissions perspective, it is therefore long-term leaching that needs to be considered. However, the present level of knowledge is regarded as prohibitive in accurately determining both the quantity and quality of future leachate emissions (Obersteiner et al., 2007). As future emissions from a landfill site cannot be measured, representation thereof requires the use of predictive models (Doka & Hischier, 2005). The development of predictive models is challenged by the complexity of landfill behaviour, necessitating the use of various assumptions and predictions, which results in a considerable level of uncertainty associated with the model output (Obersteiner et al., 2007). Given the challenge associated with the quantification of long-term leachate emissions, some LCAs assume future emissions from landfills will be negligible, thus (perhaps inaccurately) presenting landfill as a better choice than alternative waste management options (Obersteiner et al., 2007).

Not only does the choice of time frame influence the quantity of emissions, but further complicates the determination of their impact potential. Therefore, accounting for the temporal dependency of landfill emissions poses a challenge for both the LCI and LCIA stages of a LCA. In current LCA methodology, limited distinction is made with regards to temporal differences in LCI data as whole (Yuan et al., 2015). LCA typically focuses on the total mass of emitted substance (Hauschild et al., 2008), with all emissions generated over the course of a product's life cycle aggregated and treated as a single emission, generated and released at one point in time (Owens, 1997). Indeed, the default procedure for accounting for future emissions in LCA is defined in the ISO 14040:2006 standards to exclude spatial and temporal dependencies in the inventory data (ISO, 2006). When applied to landfill modelling, this approach can be contentious. Given that emissions from landfill sites typically occur at low concentrations over an extended time period, the actual impacts associated with these emissions might be lower than LCA results might suggest (Kirkeby et al., 2007), illustrating a so called "dilution in time" (Bjarnadóttir et al., 2002:77). Therefore, because LCIA results focus on the total emitted mass rather than a concentration (i.e. assigning the same impact score to a mass of emitted toxin regardless of whether this mass was emitted in a pulse or over centuries), this approach can introduce a strong bias between the landfill process and other processes in a product life cycle for which emissions typically occur over a much shorter and measurable time frame (Hauschild et al., 2008).

While the quantification of long-term emissions presents a challenge in constructing a landfill LCI, assessing the environmental impacts thereof has been raised as an important methodological debate (Hellweg & Frischknecht, 2004). Given that spatial and temporal differentiation is typically not accounted for in the LCI, historically most standard LCIA methods have not been designed to take such considerations into account (Hellweg, 2000). While the aggregation of emissions and use of a generic temporal characterisation factor in a LCIA (Potting & Hauschild, 2005) presents a simplified approach to impact modelling, this simplification could lead to large uncertainties in the final LCIA result (Yuan et al., 2015), given the potential temporal variability of the characterisation factor (Potting & Hauschild, 2005, Shah & Ries, 2009).

Environmental decision-making requiring the trade-off between present and future generations has long been recognized as a challenge in LCA modelling, giving rise to a number of ethical issues centred around intergenerational "fairness and equity" (Hellweg, Hofstetter & Hungerbuhler, 2003:8). The management of temporal issues within LCA has been the subject of on-going discussion, notably posing a key debate at the 22nd Discussion Forum on LCA in 2004. At this conference, while general consensus was reached regarding the need for the inclusion of all impacts in LCA regardless of when they are released, no consensus was reached with regards to whether these impacts should be weighted equally (Hellweg & Frischknecht, 2004). As a temporary solution for landfill representation, it was proposed that emissions be categorised according to the time frame associated with their release, with emissions occurring within the first 100 years after waste deposition classified as "short-term" and those occurring beyond that point classified as "long-term" (Hellweg & Frischknecht, 2004:339). Although no differentiation is made in the impact factor used to evaluate the impact potential of these emissions, the differentiation allows emissions and their relative impacts to be viewed as either immediate or occurring in the future.

Although various arguments have been put forward both pro and contra the separate valuation of short- and long-term emissions (i.e. Hischier et al. (2010)), limited consensus has been reached with regards to a consistent approach for managing these differences. While a comprehensive framework for the temporal discounting of future emissions has been proposed by Yuan et al. (2015) with the objective of eliminating temporal differences of inventory data, it is acknowledged that this framework can only be considered a part of the solution to addressing this issue. Remaining challenges in solving this issue include the development of standard methodologies capable of computing the temporal scale of a LCA, managing the temporally differentiated inventory data and modelling the actual environmental behaviour of these emissions (Yuan et al., 2015:30). However, it has been suggested that with accurate time dependent data, the technical application of temporal discounting is less of a challenge than addressing the "appropriateness of discounting", which remains a controversial issue (Hellweg, Hofstetter & Hungerbuhler, 2003:16).

2.4.4 Developing a Landfill Inventory

According to the *Guidelines for the use of LCA in the waste management sector* (Bjarnadóttir et al., 2002), the inventory data requirements for a landfill process are significant, with the quality of the data emphasised as having a significant impact on the LCA result. In general, there are two major approaches towards obtaining landfill LCI data: empirical results from direct measurements taken at landfill sites and experimental studies or multi-input inventory modelling tools (Obersteiner et al., 2007).

A major shortcoming in the use of empirical results is that they are only applicable to processes similar to the landfill conditions from which the data was obtained (Obersteiner et al., 2007). Using an empirical approach, data is generated by means of a "black box" principle: the emission output is directly dependent on the waste input and hence, is "incapable of reflecting changes in waste composition" (Obersteiner et al., 2007:S61).The fact that LCA requires waste type specific emissions, e.g. PET rather than MSW, is the critical shortcoming in the use of empirical results for a landfill LCI. Furthermore, unlike other waste management processes such as waste collection, recycling, and treatment, where direct data measurements are possible, determining reliable LCI data for a landfill is challenging, due to the complexity and heterogeneity of landfill sites and the time frame associated with emissions (Obersteiner et al., 2007). Experimental measurements of landfill emissions are typically undertaken by means of anaerobic digestion studies (digesters), laboratory landfill-simulator studies (lysimeters), and field-scale studies (test-cells) (El-Fadel, Findikakis & Leckie, 1997:5). However, the application of experimental results to represent full scale landfill operations is limited, as experiments are typically designed to "simulate only average landfill conditions and do not account for variations brought about by climatic and operational events" (El-Fadel, Findikakis & Leckie, 1997:5).

Multi-input inventory modelling tools by contrast, are capable of accounting for specific waste composition, and hence are preferred for their capabilities in meeting the "goal of mass balance" across the landfill system (Obersteiner et al., 2007:S62). Various modelling approaches can be used within these tools, with common approaches identified by Obersteiner et al. (2007:S61) as follows:

- **Consideration of theoretical maximum load**
  - Definition of the maximum emission potential caused by the pollutant content in waste
- **Determination of waste-specific behaviour**
  - Definition of waste degradability and release factors (to portray the effect of re-precipitation occurring in the waste body)
  - Use of laboratory tests to determine the availability of substances embodied in the waste
- **Calculation based on landfill specific parameters such as transport model or geochemical model**
  - Calculation of elemental release factors, calibrated according to field measurements
  - Consideration of uncertainties in chemical and biological interactions and preferential flow of leachate can be considered

The development of accurate predictive landfill emissions models is not without its own challenges and complexities. The heterogeneity of landfill systems, and the various complexities in the chemical and hydrological processes that occur on the site, challenge understanding and accurate representation of landfill systems (Doka & Hischier, 2005). Waste degradability is influenced by various waste-specific and site-specific parameters (Obersteiner et al., 2007). Waste-specific parameters include the waste composition, its moisture content, and density (Bhailall, 2015, Doka, 2003d), while site-specific parameters include — amongst others — site operations such as covering and compaction, pH, temperature, precipitation, and the presence of microbes and various inhibitors (Bhailall, 2015, Doka, 2003d, El-Fadel, Findikakis & Leckie, 1997). An illustration of the influence of various waste- and site-specific parameters on the key factors affecting waste decomposition is shown in Figure 2.10.

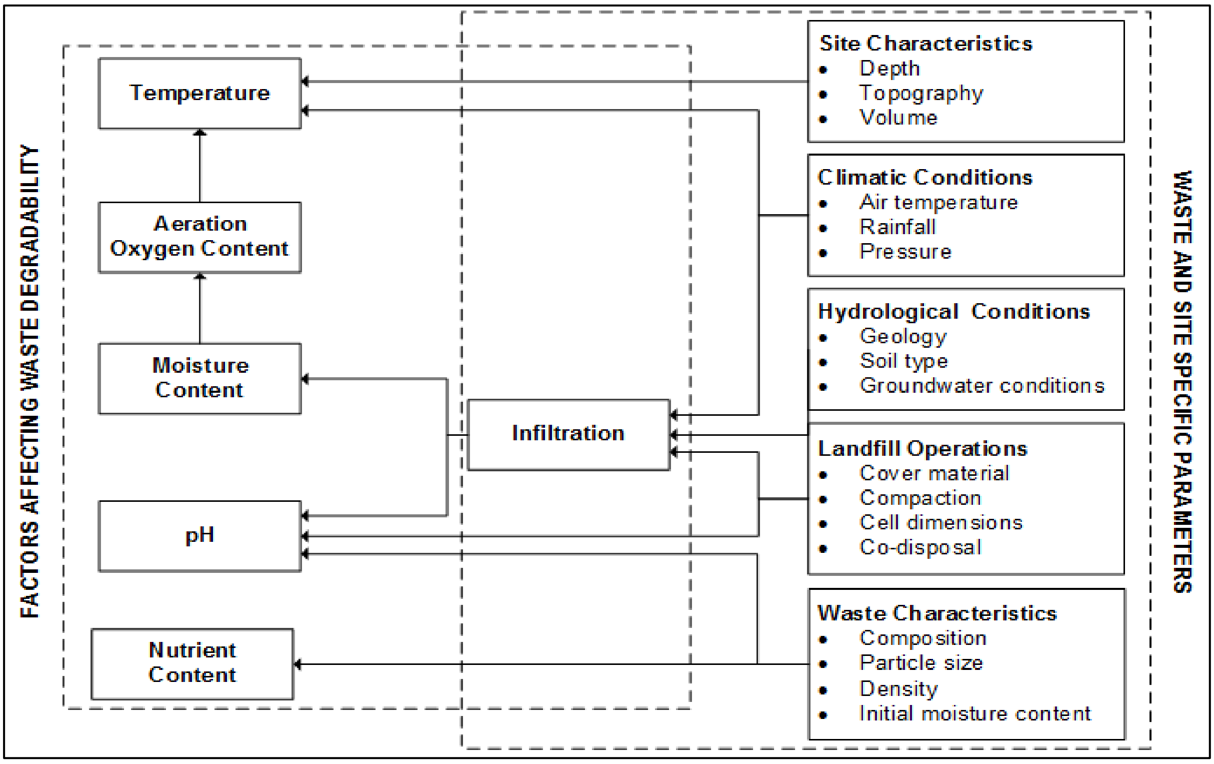

Figure 2.10 Influence of various waste- and site-specific parameters on the key factors affecting waste decomposition (Adapted from Morris (2001) as cited in Bhailall, 2015:25)

While various landfill models have been developed, major drawbacks in existing approaches have been identified as the lack of a multi-phase approach to modelling (with efforts focused on either leachate or LFG generation) and the "oscillation of approaches between extreme simplicity and overwhelming complexity" (Zacharof & Butler, 2004:454). A comprehensive multi-input modelling approach used for the development of waste-specific landfill inventories for the ecoinvent database has been presented by Doka (2003d). According to Doka (2003d), the functionality of multi-input inventory modelling in generating waste-specific emissions data requires information on both waste-specific degradation and the element specific release factor. Waste-specific degradation provides an indication of the "decomposition and mineralisation of the materials in a waste matrix" (Doka, 2003d:42), while the element specific release factor takes into account the "discrepancy between the amount of waste that is decomposed (in the waste matrix) and the actual emissions from the landfill" (Doka, 2003d:45). The elemental specific leachate emission output from landfills (E_e) can therefore be determined as a function of the mass of a specific element contained in the specific waste fraction (m_e), the waste-specific degradation (D), the element specific release factor (r_e), and the fraction the degraded element emitted via landfill gas ($\%gas_e$) as shown in Equation 2.1 (as derived by Doka (2003d:43-46)).

$$E_e = m_e \times D \times r_e (1 - \%gas_e)$$

Equation 2.1

The determination of the necessary parameters to enable the determination of waste-specific emissions can be challenging. While waste degradability can be derived from relevant literature (see Doka (2003d:43-45), the release factor is largely dependent on empirical measurements obtained from generic landfill data (Doka, 2003d, Obersteiner et al., 2007). Therefore, although the estimation of waste-specific emissions using the element specific release factor can provide "tolerable results", it is possible that several elements may be misjudged, thus limiting the application of this methodology to the entire periodic table (Obersteiner et al., 2007:S63).

An alternative approach, proposed by Zacharof and Butler (2004), uses the stochastic simulation of hydrological and biochemical processes to obtain a simplified representation of key physical, chemical, and biological processes using minimal parameterisation. To accommodate issues associated with complexities in the landfill environment and the paucity of reliable input data, this approach incorporates a formal methodology to include the effect of uncertainty in the input data on the performance of the model (Zacharof & Butler, 2004). This approach is still dependent on empirical data and hence, despite seeking to address the associated uncertainties, is not invulnerable to the limitations thereof.

The use of empirical data for the underlying assumptions in the majority of multi-input inventory approaches means that the underlying modelling assumptions will be reflective of certain landfill conditions. This represents an important limitation in the use of such models, as they do not consistently account for variations in landfill specificities such as climate, precipitation, temperature, height and type of cover layer, density, and permeability (Obersteiner et al., 2007).

The inherent variability of landfill sites limits the generation of accurate landfill inventory data by either an empirical or multi-input inventory modelling approach, with both approaches having the potential to misrepresent both current and future emissions (Obersteiner et al., 2007). This view is supported by Gentil et al. (2010), who recognise site-specific factors as prohibitive in the development of generic and harmonised models and datasets for both landfill sites and waste modelling in general. These shortcomings can manifest as inaccuracies or data gaps in the resulting inventory, limiting the inclusion of various impact categories in the LCIA or else causing them to be inadequately represented (Finnveden (1998) as cited in Obersteiner et al. (2007:S62)). In waste LCAs, it has been suggested that data gaps typically occur in human and ecotoxicological impact categories, due to the high quantity and variability of pollutants contained in waste, coupled to a lack of knowledge and limited capacity for accounting for the behaviour of these pollutants (Obersteiner et al., 2007).

2.5 Greenhouse Gas Emissions from Landfill

2.5.1 Overview of Landfill Greenhouse Gas Emissions

As discussed in Section 2.4.2, LFG emissions make a significant contribution towards the short-term impact potential of a landfill site (0 – 100 years), particularly with regards to global warming potential. To put this contribution into context, GHG emissions from post-consumer waste and wastewater contribute approximately 3% to total global anthropogenic GHG emissions (Bogner et al., 2008:16). Landfills are an important contributor within this sector, with the total CH_4 from landfills and wastewater collectively accounting for 90% of the waste sector emissions and approximately 18% of global anthropogenic CH_4 emissions in 2004 (Bogner et al., 2008:16). Consequently, LFG emissions have received special scientific and policy attention.

Mitigation potential for the waste sector typically focuses on landfill CH_4 as the baseline (Bogner et al., 2008). In developed countries, GHG emissions from landfills have largely been stabilised, following encouraged — and in some cases mandated — emission reduction by various regulations and policy strategies such as increasing landfill CH_4 recovery and reducing the landfilling of biodegradable waste (Bogner et al., 2008:19). Developing countries exhibit the converse trend, with CH_4 emissions from landfill sites set to increase from 29% (as recorded for the baseline year, 2000) to 64% by 2030 (Monni et al., 2006). The predicted increase in GHG emissions from developing countries has been attributed in part to increased use of controlled (anaerobic) landfill practices, which promote CH_4 generation, thus generating higher GHG emissions than those incurred from previous practices such as open-dumping and burning (Bogner et al., 2008). Additionally, population growth, increasing urbanisation and affluence, and expanding waste collection services in developing regions, further increase waste volumes, and hence emissions from landfill sites (Friedrich & Trois, 2011).

The contrast in landfill emission trends between developed and developing countries occurs as a result of differences in the prioritisation of waste management objectives (Brunner & Fellner, 2007). If the objectives of waste management are accepted as (1.) to protect human beings and the environment, and (2.) to conserve resources (Brunner & Fellner, 2007:234), it can be argued that for countries with affluent economies, the first of these objectives has already been achieved, leaving such countries to focus on the second objective (Brunner & Fellner, 2007). In developing countries by contrast, human health remains adversely affected by inadequate waste management practices, meaning that protection of human health is prioritised above environmental or resource protection (Brunner & Fellner, 2007).

Within developing countries, the covering of landfills represents a popular strategy towards meeting this objective (Bogner et al., 2008). However, as noted, GHG emissions from such sites exceed those incurred from previous practices such as open dumping, as an anaerobic degradation environment increases CH_4 generation (Bogner et al., 2008). While global initiatives aided by Kyoto mechanisms, such as the Clean Development Mechanism (CDM) and Joint Implementation, exist to accelerate the introduction of engineered solutions to un-engineered landfill sites (Bogner et al., 2008), progress made in the implementation of these initiatives has been hampered by various factors, including lack of financing and local capacity (Friedrich & Trois, 2011). Overall a positive trend in GHG emissions reductions in developing countries as a consequence of Kyoto mechanisms has been observed, although African countries tend to “lag behind” in this regard (Friedrich & Trois, 2011:1585).

2.5.2 Greenhouse Gas Inventories for Conventional Landfill Sites

The quantification of GHG emissions from landfill operations are frequently determined with the use of GHG emission factors (e.g. Friedrich & Trois (2013b), Gentil, Christensen & Aoustin (2009), Obermoser, Fellner & Rechberger (2009)). These factors are typically "generic and highly aggregated", determined from the average performance of a number of processes with similar characteristics, and expressed per unit of waste (Gentil, Christensen & Aoustin, 2009:703). According to the United States Environmental Protection Agency (EPA, 2006:ES-5), such factors are beneficial in supporting "waste-related decision-making in the context of climate change", enabling the calculation of GHG emissions with improved speed and transparency (Friedrich & Trois, 2013b).

GHG emission factors from the waste sector and the tools in which they are used are typically well-defined for developed countries (particularly those in the Northern Hemisphere) but lacking for developing countries (Friedrich & Trois, 2013b). Within the context of developing countries this evolution is somewhat double-sided (Friedrich & Trois). The benefit of this evolution lies in the availability of studies and tools, providing a methodological framework and reference value to countries lacking the requisite data or capacity to develop an independent framework. This view is presented alongside the caveat that the use of emissions factors and tools out of the context in which they were developed can introduce a large error margin, leading to inaccurate results (Friedrich & Trois, 2013b). Where country or region specific emissions factors are lacking, standard practice dictates the use of default values, such as those presented in the *2006 IPCC Guidelines for National Greenhouse Gas Inventories* (IPCC, 2006b). However, the error margins associated with the use of generic values is significant, ranging from 30% for countries with periodic data collection and sampling, to 60% for countries where data sampling is limited and data quality is poor (IPCC, 2006b).

Within the waste sector, GHG accounting considers emissions from three categories: operating direct emissions, upstream indirect emissions, and downstream emissions (Gentil, Christensen & Aoustin, 2009). This approach requires that the waste management system be assessed in a broad context, taking cognisance of the interaction of the system with other systems (i.e. energy systems) in the country or region (Friedrich & Trois, 2013b). It therefore follows that the quality of the resultant emission factors (or studies making use of the resulting GHG inventory such as LCA) are dependent on both the availability and quality of waste management data, and reliable data on the energy system of a particular country or region (Friedrich & Trois, 2013b). For the landfill process, the major sources of emissions within each category are as follows (as reported by Scheutz, Kjeldsen and Genti (2009:721)):

- **Upstream indirect emissions**
  - Emissions of CO_2, CH_4 and N_2O due to fuel production, electricity consumption and site infrastructure (liners and soil)
- **Operational direct emissions**
  - Fugitive emissions of CH_4, traces of NMVOC, N_2O and halogen-containing gases
  - Biogenic CO_2 and CH_4 from waste decomposition
  - Fuel combustion from machinery (CO_2, CH_4, N_2O, traces of CO and NMVOC)
  - Emissions from leachate treatment plant (CO_2 biogenic, CO_2, CH_4, N_2O)
- **Downstream indirect emissions**
  - Production of heat and electricity from CH_4 combustion substituting fossil energy
  - Carbon bound in landfill (100 years)

The CH_4 generated from the anaerobic degradation of waste inside the landfill body represents the "prime GHG from landfilling", with the output volume dependent on both the biogenic carbon content of the waste and the degradability of the material containing the carbon (Manfredi et al., 2009a:827). Similar approaches for the determination of an emission factor for the GHG emissions arising from the waste body have been presented in detail by both the IPCC (2006b) and Manfredi et al. (2009a). An overview of this approach and further discussion is available in Appendix B, Section B.1.

In addition to CH_4 and CO_2, other GHG emissions from landfills include NMVOCs, N_2O and other nitrous oxides. These are typically considered insignificant in landfill GHG accounting, and hence excluded from standard methodological approaches (IPCC (2006b), Friedrich & Trois (2013b)). Furthermore, studies investigating the contribution of N_2O and other halogens towards landfill GHG emissions are lacking (Friedrich & Trois, 2013b), thus limiting potential for their inclusion in a GHG inventory.

Further emissions arising from the landfill process include those process-linked burdens arising from the construction and operation of the site. Although the associated impacts from these emissions are typically lower than direct emissions from the waste (Doka, 2003d), their inclusion is important for comprehensive GHG accounting. Intuitively, this data will be site specific, however, typical ranges for process-linked GHG emission sources for conventional landfilling are shown in Table 2.7.

Table 2.7 Consumption data for direct and indirect emissions arising from the construction and operation of a conventional landfill site (Adapted from Manfredi et al., 2009a:827)

Process/Emission	Description of Quantity	Consumption[a]	Reference
Provision of diesel fuel	Fuel provided for soil works at the site for construction of the landfill	0.5 – 1 L diesel.tonne^{-1}	Manfredi et al. (2009a)
Combustion of diesel fuel[b]	Fuel consumed on the landfill site dependent on the degree of compaction of waste and amount of soil excavated and/or moved for daily cover	1 – 3 L diesel.tonne^{-1}	Manfredi and Christensen (2009), Niskanen et al. (2009), Hunziker & Paterna (1995)
Provision of electricity	Electricity provided for general lighting on site, administration buildings, pipes, and fans	2 – 12 kWh.tonne^{-1}	Niskanen et al. (2009), Hunziker & Paterna (1995)
Provision of HDPE for synthetic liner	Provision of 2mm thick HDPE liner for 20m deep landfill	1 kg liner.tonne^{-1}	Manfredi et al. (2009a)
Provision of gravel	Gravel/crushed rock for construction of drainage system etc.	0.1 tonne.tonne^{-1}	Manfredi et al. (2009a)

[a] Reported as specific demand per tonne of wet waste landfilled
[b] Cumulative throughout the entire lifetime of the landfill

In recognition of the importance of the energy system applicable to a particular site in developing a GHG inventory, a comprehensive study was undertaken by Fruergaard, Astrup and Ekvall (2009), which focused on the energy use and energy recovery in waste management. The objective of this study was to provide instruction on how these aspects should be addressed consistently for GHG accounting. However, this study is limited to the provision of GHG emission data for "energy flows" i.e. emissions data for the most common fuels, electricity and heat required by waste management processes (Fruergaard, Astrup & Ekvall, 2009:725). Therefore, energy flows such as those consumed during material production were excluded from the focus of the study (Fruergaard, Astrup & Ekvall, 2009:725). A summary of the results of this study is shown in Table 2.8 overleaf. To overcome limitations in the results for accounting for the emission factors associated with the production and provision of material flows, Manfredi et al. (2009a) supplemented these findings with data from the EASEWASTE database.

Table 2.8 GHG emission factors for energy systems linked to landfill operations (As presented in Manfredi et al., 2009a:827)

Type of process/emission	Emission factor	Reference
Provision of diesel fuel	0.4 – 0.5 kg CO_2 eq.L^{-1} diesel	Fruergaard, Astrup and Ekvall (2009)
Combustion of diesel fuel	2.7 kg CO_2 eq.L^{-1} diesel	Fruergaard, Astrup and Ekvall (2009)
Provision of electricity	0.1 – 0.9 kg CO_2 eq.kWh^{-1}	Fruergaard, Astrup and Ekvall (2009)
Provision of HDPE for synthetic liner	1.85 kg CO_2 eq.kg^{-1} HDPE	EASEWASTE database
Provision of gravel	1.4 kg CO_2 eq.$tonne^{-1}$ wet waste	EASEWASTE database

The range in emission factors in Table 2.8 represents the range identified by Manfredi et al. (2009a) as most applicable to the conventional landfill systems being represented within their study. Analysis of the findings presented by Fruergaard, Astrup and Ekvall (2009) (covering waste management systems as a whole) show the range in emission factors associated with electricity to be substantially higher than that suggested by Table 2.8. For example, the GHG emission factor for electricity provision shows significant variation between countries, ranging from 0.007 – 1.13 kg CO_2 eq.kWh^{-1} (Fruergaard, Astrup & Ekvall, 2009:731). This variation was attributed to the different fuels used in the provision of electricity in different countries, variations in the penetration of renewable energy options, as well as the variable production efficiencies (Fruergaard, Astrup & Ekvall, 2009).

2.5.3 Greenhouse Gas Inventories for Unmanaged Landfill Sites

The discussion presented in Section 2.5.2 focuses predominantly on the GHG emissions arising from the operation of conventional, managed landfill processes. However, non-engineered landfills and open dumps remain prominent disposal mechanisms in many parts of the world, especially in developing countries (see Figure 2.4 and Table 2.3). Such sites typically lack any form of engineering benefit or control, such as waste compaction and covering, or control systems to collect and manage the emission output from the site (Manfredi et al., 2009a). The lack of engineered controls can, from one perspective, be considered a simplification in the determination of a GHG inventory from the site, as there is little need to inventory emissions from process-linked burdens.

Emissions from the degradation of the waste itself remain an important emission to account for. The lack of control measures essentially removes the buffer between the waste body and the environment, which implies that LFG emissions will be emitted directly into the atmosphere, and leachate directly into the soil and groundwater reserves beneath the landfill site (Manfredi et al., 2009a). Biodegradation of organic material in particular is strongly dependent on the conditions of the disposal site, occurring either aerobically or anaerobically under different conditions (Frøiland Jensen & Pippatti, 2001). In contrast to conventional sanitary sites where anaerobic or semi-aerobic conditions typically prevail, unmanaged sites have a larger proportion of waste decomposing aerobically (IPCC, 2006b), thus reducing CH_4 emissions from the site. The extent to which aerobic decomposition occurs is dependent on the specific conditions of the site, influenced by factors such as the extent of covering and compaction of the waste, disturbance of waste layers due to the presence of scavengers, depth of the site, and the height of the water table (IPCC, 2006b).

The effect of site structure and management practices on CH_4 generation has been addressed by the IPCC (2006b) with the development of a methane correction factor (MCF) reflective of different landfill conditions. The MCF accounts for the fact that well-managed sanitary sites produce more CH_4 from a given amount of waste than unmanaged sites where conditions supporting aerobic degradation prevail. MCFs have been developed for four different landfill categories, with the inclusion of a fifth category intended to be used in the case where categorisation of landfill into the predefined categories is not possible (IPCC, 2006b). An overview of the different landfill types characterised by the IPCC (2006b) and their corresponding MCFs is shown in Table 2.9.

The MCF can be applied to correct for the effect of site conditions on landfill CH_4 emissions as shown in Equation 2.2 (as presented in Friedrich and Trois (2013b:1020)).

$$CH_{4_{emitted}} = CH_{4_{generated}} \times MCF$$

Equation 2.2

The application of the MCF as shown in Equation 2.2 is consistent with the methodology outlined in IPCC (2006b). Further discussion on the use and limitations on the MCF are available in Appendix B.3.

Table 2.9 IPCC Landfill site classification and corresponding methane correction factors (Adapted from IPCC, 1997:6.8 and IPCC, 2006b:3.14)

Type of Site	Characteristics of Site	Methane Correction Factor	Uncertainty
Managed — anaerobic	• Controlled placement of waste • Degree of control of scavenging • Degree of control of fires • Will include at least one of the following: - Cover material - Mechanical compacting - Waste levelling	1.0	-10%, +0%
Managed — semi-aerobic	• Controlled placement of waste • Degree of control of scavenging • Degree of control of fires • Will include at least one of the following: - Cover material - Mechanical compacting • Waste levelling • Will include all of the following structures for introducing air to waste layer - Permeable cover material - Leachate drainage system - Regulating pondage - Gas ventilation system	0.5	±20%
Unmanaged — deep	• Does not meet criteria of managed facility • Depth > 5m and/or • High water table at near ground level (i.e. filling of a pond, river or wetland by waste)	0.8	±20%
Unmanaged — shallow	• Does not meet criteria of managed facility • Depth < 5m	0.4	±30%
Uncategorised	• No specific characteristics, to be used where countries cannot categorise their landfills into the above categories	0.6	-50%, +60%

2.5.4 Landfill Greenhouse Gas Accounting in South Africa

According to the *GHG National Inventory Report* (Department of Environmental Affairs [DEA], 2014a:68,236), in 2010 total emissions from the waste sector accounted for 3.6% of South Africa's GHG emissions, of which solid waste disposal contributed the vast majority (82.75%). However, the overall accuracy of this figure is questionable. According to this report, limitations in landfill characterisation and inconsistent and inaccurate data on the quantity and composition of waste were key contributors to the overall uncertainty of the result. Only GHG emissions generated from managed disposal sites in South Africa were documented and even for these, limitations in the accuracy of data were acknowledged (DEA, 2014a). Even for countries in which waste generation data is collected on a regular basis, uncertainty in the emissions output can be as high as 30% (DEA, 2014a), suggesting that this uncertainty is likely to be even higher in South Africa.

In recognition of the global imbalance in emissions factors and the potentially large error margin associated with their use in an alternative context to which they were developed, a study was undertaken by Friedrich and Trois (2013b) with the objective of developing GHG emission factors representative of municipal waste management processes currently undertaken in the South African waste sector. For the determination of emissions from landfills, the variation in the performance and infrastructure of different sites necessitated the development of four generic South African landfill scenarios. It was assumed that these scenarios could be considered broadly representative of landfill disposal in South Africa and were defined as follows (Friedrich & Trois, 2013b):

- Scenario 1: Landfill disposal in an open dump
- Scenario 2: Landfill site without LFG collection
- Scenario 3: Landfill site with LFG collection and flaring
- Scenario 4: Landfill site with LFG collection and electricity generation

The methodological approach taken for the determination of these factors was consistent with standard methodologies and considered direct and indirect emissions consistent with those outlined by Scheutz, Kjeldsen and Gentil (2009) (as discussed in Section 2.5.2). A number of limitations in the availability of representative and accurate data for all emission sources was acknowledged (Friedrich & Trois, 2013b). The results of this study found GHG emissions to vary across the different landfill scenarios investigated. For wet waste, accounting for carbon storage, results ranged from -145 to 1016 kg CO_2 eq.tonne^{-1}, and without carbon storage, the range was from 441 – 2532 kg CO_2 eq.tonne^{-1} (Friedrich & Trois, 2013b:1023). In both cases, landfill scenarios without LFG collection had higher emissions than those with LFG collection and/or utilisation. This result was highlighted given the observed dominance of sites without LFG collection in South Africa.

In terms of representing process-linked burdens (i.e. fuel and electricity consumption linked to on-site operations), similar limitations to those associated with sourcing waste composition data were observed, such as omissions and inconsistencies in data sources (Friedrich & Trois, 2013b). A comparison of the process-linked GHG emissions applicable to South African landfill conditions to values reported in literature is shown in Table 2.10 overleaf. Comparison of these results to those presented for the stored carbon, CH_4, biogenic CO_2 emissions and direct GHG emissions from South African landfill scenarios (as determined by Friedrich and Trois (2013b)), suggests that the GHG emissions arising from indirect emissions are significantly lower than the direct emissions associated with the decomposition of the landfilled waste. This result is consistent with those presented by Manfredi et al. (2009a) for the GHG emissions arising from European landfills operations.

Table 2.10 Comparison of process-linked GHG emissions for South African landfill sites with relevant literature values (Adapted from Friedrich & Trois, 2013b:1024)

GHG emission type	Amount used per tonne of wet waste from literature (Manfredi et al., 2009a)	Amount used per tonne wet waste for local landfill sites	kg CO_2 eq per tonne of wet waste for local calculations[a]
Diesel for construction of cells on landfill site	0.5 – 1 L	No records for South Africa	2.7[c]
Synthetic liner for construction of cells (HDPE, 2mm)	1 kg (assuming a cell depth of 20 m)	No records for South Africa	1.9
Provision of sand, gravel or crushed rock needed on site	0.1 tonne	0.4 tonnes[d]	0.2
Electricity for on-site lighting, administration buildings, pumps and fans	2 – 12 kWh[b]	0.9 kWh[e]	2.03[f]
Diesel for daily on-site operations	1 – 3 L	0.34 L[g]	2.7

[a] Associated emissions factors used in calculation from Manfredi et al. (2009a)
[b] Niskanen et al. (2009) in Manfredi et al. (2009a)
[c] Higher value from literature used (worst case scenario)
[d] Local value obtained as an average over three years from three eThekwini Municipality landfill sites
[e] Local value obtained as an average over one year from one eThekwini Municipality landfill site with leachate and gas collection systems
[f] Local electricity factor used
[g] Local value obtained from one landfill site over 1 year

In line with increasing recognition of the contribution of LFG emissions towards global CH_4 emissions, a quantification of LFG emissions at a national level has been presented by Bhailall (2015). According to the results of this study, in which actual LFG yields were compared to various theoretical landfill emission models[4], the use of such models to predict landfill emissions is limited within the South African context. One of the short-comings in the use of predictive models in simulating South African landfill conditions was attributed to their inability to take into account site-specific characteristics such as covering material and thickness, climatic conditions and seasonal variability in CH_4 oxidation rates, and daily site operations (i.e. covering and compaction) (Bhailall, 2015). As the majority of research into predicting LFG emissions has been concentrated in the Northern Hemisphere, the development of the required input parameters to these models are typically not representative of the conditions that exist in Southern Africa (Fourie & Morris, 2004). This implies that for an accurate quantification of emissions at a national level, the effect of site-specificities on LFG emissions should be accounted for.

The results presented by Bhailall (2015) further highlight the importance of good quality data to use as an input into predictive models. In South Africa, obtaining high-quality data presents a key challenge, with a significant number of operational landfills reportedly keeping “minimal to non-existent” data records on the operation of the site (Bhailall, 2015:81). Furthermore, the availability of waste characterisation and waste tonnage data was also found to be limited, with the latter being attributed to the lack of infrastructure such as weighbridges at the majority of landfill sites (Bhailall, 2015). Without accurate and reliable data from landfill sites, the quantification of LFG emissions and their relative contribution towards global warming will remain limited (Bhailall, 2015, Friedrich & Trois, 2013b).

[4] Theoretical models used were LandGEM, GasSim, IPCC Waste Model 2006 and the California Landfill Methane Inventory Model (CALMIM) (Bhailall, 2015)

2.6 Leachate Emissions from Landfill

2.6.1 Overview of Landfill Leachate Emissions

In addition to LFG emissions, leachate emissions are also a significant contributor towards the short-term impact potential of a landfill site. As discussed in Section 2.4.2, from a LCA perspective, the short-term impact potential of leachate is typically seen in spoiled groundwater resources and ecotoxicity impact categories. From a long-term perspective, given the inevitable failure of barrier systems and controls, for slowly degrading waste types, leachate emissions can continue to occur for centuries or millennia beyond the operational phase of the landfill (Doka & Hischier, 2005). Consequently, long-term landfill leachate emissions can present a chronic environmental problem.

Landfill leachate is defined as "the aqueous effluent generated as a consequence of rainwater percolation through wastes, biochemical processes in waste's cells and the inherent water content of wastes themselves" (Renou et al., 2008:469). Water therefore plays an integral role in the leachate generation potential of a landfill, with its distribution and transport key parameters in this regard (Fellner & Brunner, 2010). A simplified schematic illustrating the generation and release of emissions from a landfill site is shown in Figure 2.11.

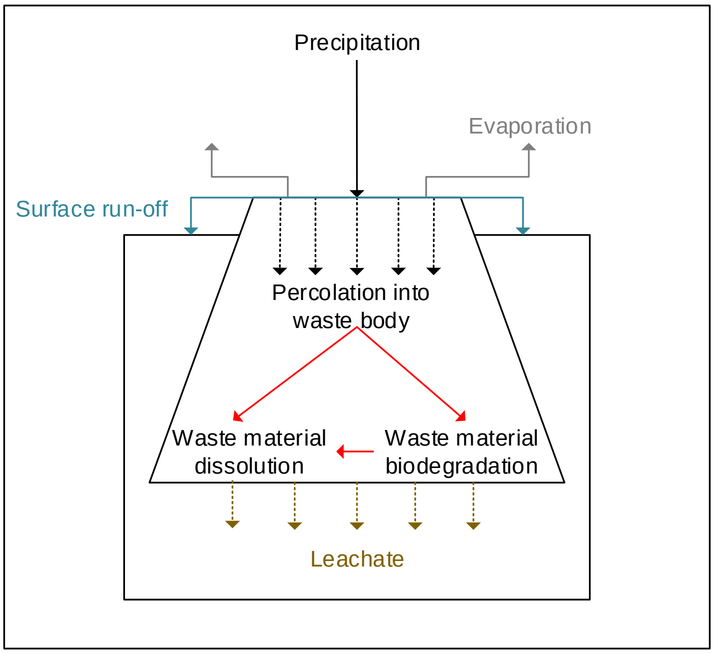

Figure 2.11 Schematic diagram showing the generation of leachate in a landfill site (Adapted from Kostova (2006) as cited in IRMA, 2016:258)

Managing leachate has been highlighted as one of the most significant challenges facing the design and operation of landfill sites (Information Resources Management Association [IRMA], 2016). Although highly variable, constituents typically encountered in leachate are classified into three main groups: 1.) organic matter (i.e. volatile fatty acids and humic-type constituents), 2.) inorganic matter (i.e. ammonia, nitrogen, phosphorous, inorganic salts), and 3.) heavy metals (IRMA, 2016, Renou et al., 2008).

2.6.2 Modelling Leachate Emissions

While a wide range of methods exist for predicting landfill leachate (Fellner et al., 2009), it has been suggested that amongst the most widely used are the *Hydrologic Evaluation of Landfill Performance Computer Method* and the *Water Balance Method* (IRMA, 2016:258-259). The *Water Balance Method* in particular is used extensively in the determination of leachate quantities (IRMA, 2016). Whilst on-site field measurements can also be undertaken, comparison of these results with laboratory models have found results to be inconsistent (Fellner et al., 2009). Furthermore, field measurements will yield site specific results, which due to the variability of landfill site conditions, limits the usefulness of this method in representing a general case. The determination of leachate emissions by means of a modelling approach requires a number of parameters including the net water input (i.e. balance of precipitation less evapotranspiration and surface run-off), surface and groundwater infiltration rate, waste composition and state, surface liquid storage, surface area, and site geology (IRMA, 2016).

Therefore, like LFG emissions, leachate emissions are dependent on both climatic factors and other site-specific parameters. Climatic factors influence both precipitation and evaporation, and hence affect the water balance of the site (Obersteiner et al., 2007, Renou et al., 2008). Additional factors that influence leachate quantity and concentration include the nature of the waste (toxic potential, water content, and degree of compaction), site infrastructure and operating procedures, and age of the landfill site (Obersteiner et al., 2007, Renou et al., 2008). The composition of leachate in particular is highly variable, and is strongly dependent on a variety of factors including the waste type (IRMA, 2016).

Given that different landfill sites reflect a high level of variation in both climatic and other site-specific parameters (Hellweg, 2000, Obersteiner et al., 2007, Renou et al., 2008), and the sensitivity of leachate to these parameters, developing a general case for pollutant release from the waste body is complex (Hellweg, 2000). It cannot, for example, be assumed that all leachate emissions will immediately enter the surface water — rather, it should be assumed that emissions will seep into the subsoil before they eventually reach the groundwater beneath the site (Hellweg, 2000). Accounting for the time delay of leachate emissions in the subsoil, in addition to the time scale associated with waste decomposition and subsequent leachate release (as discussed in Section 2.4.3), is particularly challenging within the context of a LCA (Hellweg, 2000). Although these challenges have been investigated in depth by Hellweg (2000) and Doka (2003d), these are not going to be reviewed in detail here.

For the purposes of LCA, although it is desirable to generate inventory data reflective of both the geographical and temporal scope defined for the study, generating such an inventory for leachate emissions by "simple recipe like models" is not seemingly possible without a number of simplifying assumptions (Obersteiner et al., 2007:S65). Furthermore, as the environmental impacts associated with landfill leachate are often localised, this poses a challenge to the LCIA stage, as LCIA approaches are typically "site-generic", thus not allowing any "environmental sensitivity" to be expressed (Kirkeby et al., 2007:970). Consequently, determining the impact potential from landfill sites is recognised as an ongoing challenge within LCA (Doka, 2003d, Hauschild et al., 2008, Hellweg & Frischknecht, 2004, Hellweg, Hofstetter & Hungerbuhler, 2003).

Landfill leachate is increasingly being recognised for its potential for pollution (Butt et al., 2014) and consequently has received attention as an environmental protection concern. Unlike GHG emissions, however, where developing national inventories is part of an international agenda, quantification and documentation of landfill leachate emissions at both national and international level remains limited. Consequently, the availability and/or accessibility of predictive models and methodologies for developing inventories for leachate emissions for such applications as LCA are lacking. This is not to say that there is no action being taken towards monitoring and mitigating leachate related impacts. Stricter environmental requirements are continuously being imposed for the protection of ground and surface waters, driving the development of leachate treatment processes capable of reducing the pollution potential of landfill leachate (Renou et al., 2008).

2.7 Summary of Literature Review

While LCA is a recognised tool for improving the sustainability of consumption and production patterns by providing a quantitative basis for incorporating LCT and LCM into product design, its application is limited in representing the end-of-life stage. Historically, this stage has received limited attention in comparison to other stages in a product's life cycle (Bjarnadóttir et al., 2002). There exists a notable imbalance in the geographical distribution of studies, with the majority concentrated in Europe with little representation of waste management systems in developing countries, suggesting a poor penetration of LCA into these regions (Laurent et al., 2014a). This historical and geographical imbalance, however, is being redressed as the end-of-life stage is increasingly recognised for its "far from negligible" (Obersteiner et al., 2007:S69) contribution towards the overall impacts of a product's life cycle.

The historical paucity in end-of-life modelling can be attributed to various factors, such as the complexity of waste systems, and limitations in the availability of both waste- and site-specific data. These challenges are frequently encountered in the representation of landfill disposal, a waste management practice that still dominates in terms of global waste management. Modelling the landfill process within the context of LCA is further complicated by the associated time frame for emissions, which can continue to occur for centuries or millennia beyond the initial deposition of waste (Doka & Hischier, 2005). Although the management of temporal issues within LCA has been the subject of on-going discussion, limited consensus has been reached regarding how to evaluate future impacts, and whether present and future impacts should be weighted equally (Hellweg & Frischknecht, 2004).

Modelling landfill disposal as an end-of-life management option is particularly relevant in developing countries. However, landfill practices in developing countries (unmanaged landfills and open dumps) can vary from those characteristic of developed countries (sanitary landfill). Although various models and datasets exist to inventory landfill emissions, the geographic specificities of landfill operations means that models and datasets are typically optimised for a specific geography and are consequently best representative of landfill processes occurring within the same geographical context (Gentil et al., 2010). Existing models and datasets are concentrated within the developed countries of the Northern Hemisphere. The variation in landfill practices that occurs between the developed and developing world is therefore prohibitive in the uptake of these models/datasets in developing countries.

Developing a suitable model or inventory for landfill disposal in the context of a developing country is challenging. Developing countries typically lack waste data and obtaining suitable landfill inventory data is difficult. Unlike other waste management processes such as waste collection, recycling, and treatment, where data can be obtained by direct measurements (Obersteiner et al., 2007), landfill emission data is both site and time dependent, making both experimental measurements and measurements from landfill sites themselves difficult to perform. LFG has received increasing recognition for its contribution towards global GHG emissions, prompting the development of emission factors to quantify this contribution at both national and global levels. In South Africa, national efforts to inventory GHG emissions has resulted in focused studies directed towards quantifying GHG emissions from the waste sector (i.e. Bhailall (2015), Friedrich & Trois (2013b)). Unlike GHG emissions, where developing national inventories is part of an international agenda, quantification and documentation of landfill leachate emissions at both national and international level remains limited.

Chapter 3

RESEARCH METHODOLOGY

This chapter describes the research approach and methodology used to meet the objectives laid out for this dissertation. It starts by introducing the research questions that guided the study and thereafter provides detail on the specific approach and research methods undertaken in each research stage.

3.1 Research Questions

Pertinent findings from the literature review not only confirm that waste management practices vary between developed and developing countries, but further illustrate the relative difference in environmental impacts that can be incurred by different waste management options. Where waste management practices in developing countries differ from those in developed countries (which are typically well represented within existing LCA capacity), it has been suggested that the generation of a representative dataset is limited by, amongst else, a lack of scientific capacity, and accurate and representative waste data.

In order to ascertain the applicability of this statement within the South African context, it was of relevance to investigate the extent to which end-of-life practices in South Africa can be represented with existing LCA datasets and where these fall short, to what extent they can be modified to better represent reality. This investigation therefore required an understanding of current South African waste management practices in addition to a review of LCA capabilities in representing the status quo. To inform this investigation and meet the objectives outlined for this project, three broad research questions were defined:

1. What is the status quo for general waste management in South Africa?
2. How applicable is current LCA capacity for representing end-of-life in South Africa?
3. To what extent can LCA models and datasets be modified to represent alternative end-of-life practices?

These questions provided the framework for the research, establishing the need for a sequential approach towards meeting the objectives laid out for this dissertation. Each research question was therefore regarded as a separate stage, with the research approach for each stage informed by directed key questions. The key questions defined for each research stage are as follows:

Stage 1: Establish the Status Quo for General Waste Management in South Africa

i. What are the current waste management practices and "market share" of each for municipal waste in South Africa, and do they vary significantly from those in developed countries?

Stage 2: Review Current LCA Capabilities for Representing Product End-of-Life in South Africa

ii. To what extent are South African waste management practices represented by existing LCA datasets?
iii. What are the main parameters in existing datasets that are influenced by local conditions?
iv. To what extent can existing datasets be adapted to represent local conditions?
v. What additional considerations should be incorporated into LCA end-of-life modelling to better represent local practices?

Stage 3: Approximate Product End-of-Life in South Africa by Means of Dataset Modification

vi. To what extent does modification of existing LCA datasets improve representation of local practice?
vii. What are the important parameters in the representation of local waste management practices that affect the LCIA result and where should focus be placed in terms of representing alternative waste management scenarios?

The development of the methodological approach to address the key questions in the latter research stages was informed by the outcomes of the previous stages. For example, to address key questions vi – vii, it was necessary to understand the availability and parameterisation potential of existing LCA models and datasets (key questions ii – v). The review of LCA capabilities in representing South African waste management practices was dependent on knowledge of the status quo with regards to waste management (key question i). Therefore, the three main stages for this research were undertaken sequentially, with the outcomes of each stage used to inform the development of each subsequent stage. An overview of the methodological approach is shown in Figure 3.1.

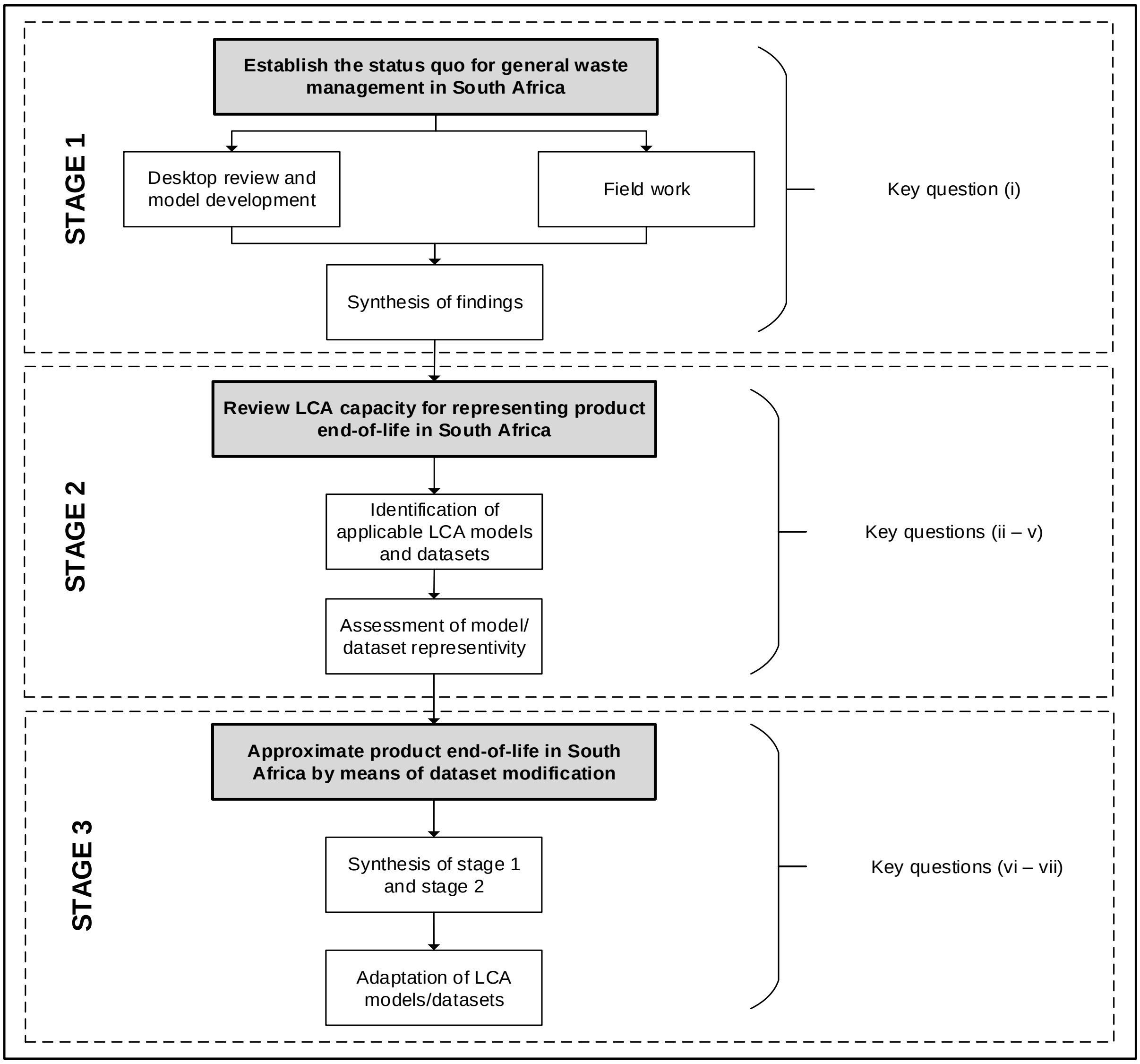

Figure 3.1 Overview of the methodological approach to address the key questions and meet the objectives defined for the research

3.2 Methodological Approach: Stage 1

3.2.1 Overview of Approach

The first stage of the research aimed to address the first key question and establish the status quo with regards to waste management in South Africa. This was undertaken in two parts: a desktop review incorporating both a review of government and other publicly available reports applicable to waste management and the development of a model to quantify waste flows, and a field work component in which stakeholders within the waste management industry were interviewed and waste management facilities visited. The focus of this investigation was limited to general (municipal) waste with a specific focus on domestic waste. The results from these two parts were then synthesised with literature findings to address the key questions posed for this section. This approach is shown schematically in Figure 3.2. Further detail regarding the nature of the work undertaken in each component is discussed in the following sub-sections.

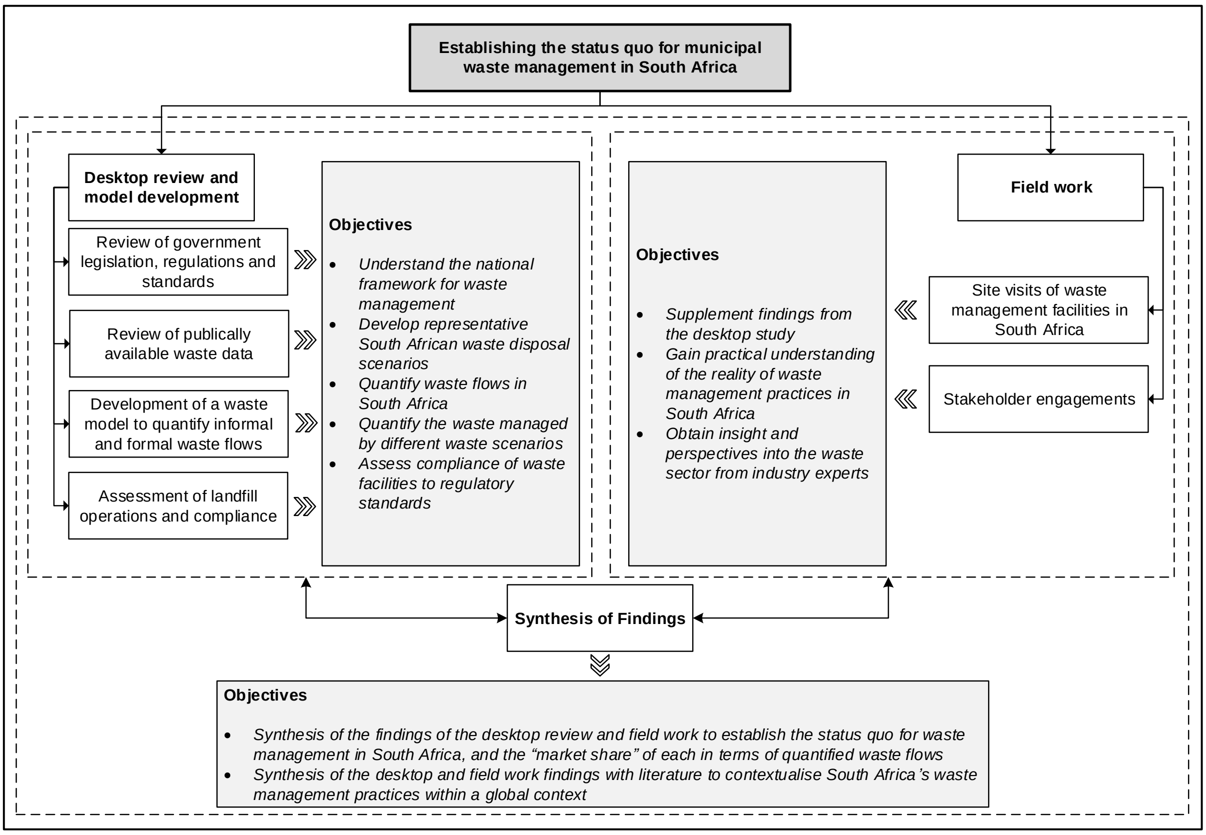

Figure 3.2 Schematic illustration of the methodological approach to establish the status quo for general waste management in South Africa

3.2.2 Desktop Review and Model Development

The objective of the desktop review and model development was to interrogate the availability of waste data and the current state of knowledge within the waste sector, and develop a quantified estimate of waste flows so as to determine the "market share" of different waste management practices utilised in South Africa. The approach taken to meet this objective can be broadly split into two parts: establishing the national standards for municipal waste management through a review of relevant government legislation and waste policy and then comparing these with the actual municipal waste management practices occurring in the country through a review of publicly available waste reports, municipal waste data and landfill audit reports, and the development of a model to estimate waste flows.

3.2.2.1 Review of National Waste Legislation

A review of national waste legislation was undertaken to establish the national standards with regards to waste management in South Africa. The focus of this review was to understand the national framework for waste management and the key legislation and standards within this framework. This review was not intended to provide an exhaustive summary of waste legislation, but rather to emphasise important regularly standards and requirements for the management of municipal waste. Documents reviewed during this investigation are available in Table C.1 (Appendix C, Section C.1).

3.2.2.2 Waste Quantification and Characterisation of Waste Flows

Various sources of waste data were interrogated in order to assess the current state of knowledge and availability of data within the waste sector. Four key sources of waste data capable of representing waste flows in South Africa were identified. These are as follows:

- The South African Waste Information System (SAWIS) (as available through the South African Waste Information Centre (SAWIC))
- The *National Waste Baseline Report* (DEA, 2012a)
- Waste generation rates as reported by the IPCC (2006a), Hoornweg and Bhada-Tata (2012), DEAT (2006), Department of Agriculture Conservation and Environment North West Provinical Government (2008), Gauteng Department of Agriculture and Rural Development (2009), Ogola, Chimuka and Tshivhase (2011), Fiehn and Ball (2005) and the Western Cape Government Department of Environmental Affairs and Development Planning (2010) (as quoted in Myers and Pieterse (2014))
- Integrated Waste Management Plans (IWMPs) as compiled by local government

Each data source was reviewed with regards to its capability to quantify waste flows in South Africa. Given the variation in the time period considered in each study, where necessary, data was extrapolated to 2016 using population statistics so as to provide a consistent basis for comparison. This approach is shown in Equation 3.1.

$$\text{Total waste quantity in 2016} = \frac{\text{Waste quantity in year x}}{\text{Population in year x}} \times \text{Population in 2016}$$

Equation 3.1

Having reviewed these sources of waste data, limitations in their capabilities in accounting for informal waste were identified. In order to address this shortcoming, a model was developed to estimate the quantity of informal household waste generated and disposed of in South Africa. This model was based on information contained within the *General Household Survey 2015* Report (Stats SA, 2016a) and combined with national waste generation rates to obtain quantified estimates for the quantity of waste generated and disposed of informally in South Africa. Further detail on the development of this model is provided in Chapter 4, Section 4.4.

3.2.2.3 Assessment of Landfill Operations and Compliance

According to the findings from the literature review (Chapter 2), both developed and developing countries retain a strong dependence on landfill. The results from the desktop review (as detailed in Chapter 4) confirmed that South Africa displays a similar dependence on landfill disposal. Based on this observation, a detailed investigation was undertaken with the objective of identifying the relevant regulations, operating standards and conditions applicable to landfill disposal in South Africa. While a review of specific legislation provides an indication of the legal requirements for landfill sites, to determine the reality of landfill operations, a review of 66 external audit reports conducted on licensed waste disposal facilities within the Western Cape over the course of the 2015/2016 year was undertaken. These reports were made accessible by the Western Cape Government Environment Affairs and Development Planning Department of Waste Management. The objective of this review was to determine how closely landfill facilities comply with their license requirements and where — or indeed whether — non-compliance occurred with an appreciable frequency.

3.2.3 Field Work

The objective of the field work component was to supplement the information obtained through the desktop study through a combination of on-site visits and stakeholder engagements with various role-players in the waste sector.

3.2.3.1 Site Visits

Various site visits were untaken with the objective of gaining practical understanding of, and insight into, the reality of waste management in South Africa. The site visits involved an extensive tour of the facility and informal interview with the facility manager or site operator. A list of the waste facilities visited is shown in Table C.2 (Appendix C, Section C.1).

3.2.3.2 Stakeholder Engagement

This component of the research involved engagement with waste facility managers, operators and other stakeholders and experts within the waste management sector. It was intended that the stakeholder engagement process supplement the findings from the desktop review as opposed to providing a primary source of data for the study. The stakeholder engagement was undertaken through a semi-structured interview approach: interview questions were prepared but only served as a basis for guiding the discussion. A list of stakeholders and main points of discussion in each interview is available in Table C.3 (Appendix C, Section C.1).

The objective of the interview process was to supplement the findings of the desktop review and gain further insight into the nature of waste management in South Africa. The nature of questions posed related specifically to:

- The extent to which waste management facilities are compliant with waste legislation
- Major challenges facing waste management facilities and the waste sector in general
- The current uptake of alternative waste management strategies in South Africa
- The future outlook regarding the uptake of alternative waste management strategies

3.2.4 Synthesis of Findings

The synthesis of findings was intended to tie together the findings from the desktop review and field work component of the research to establish the status quo for waste management in South Africa. It should be noted that although represented as a separated component of the methodology, the synthesis of findings was integrated throughout the desktop and field work components and as such, is not presented as a separate section in the results.

3.3 Methodological Approach: Stage 2

3.3.1 Overview of Approach

The definition of representative waste management practices undertaken in South Africa provided the basis for Stage 2: a review of LCA capabilities for representing product end-of-life in a South African context, informed by key questions ii – v. The approach to answering these key questions was undertaken in a series of steps, which are outlined below.

1. Definition of representative waste management practice/s in South Africa (informed by the findings of Stage 1 of the methodology).
2. Identification of representative datasets within SimaPro v8.3 for the waste management practice/s defined in Step 1.
3. Assessment of the datasets identified in Step 2 with regards to their application to South African conditions.
4. Investigation into the parametrisation potential of existing datasets to facilitate improved representation of local conditions with regards to end-of-life LCA modelling.

It should be emphasised that the objective of this stage of the research was to investigate current LCA capacity with regards to modelling product end-of-life in a South African context. It was not the express intention of this research to develop a representative dataset for South African conditions, but rather to investigate the possible adaptations that can be made to existing datasets to improve their application to local conditions. It was intended to identify practical adaptations that could feasibly be undertaken by a product developer or LCA practitioner in lieu of representative end-of-life datasets for local conditions. Therefore, adaptation potential was assessed with regards to the practicality of its application.

3.3.2 Definition of Representative Waste Management Practices

Although various options exist for the management of general waste in South Africa, the interrogation of LCA capacity for all possible practices is unfeasible. Therefore, it was necessary to define representative practises to be investigated in further detail. This was based on the findings from the status quo analysis (Chapter 4), which indicated that waste management in South Africa is dominated by landfill disposal with conditions ranging from well-managed sanitary landfills to open dumps.

Therefore, the interrogation of datasets was limited to those providing an inventory for waste disposal to land (either formal landfilling or open dumping). This decision was not only based on the strong dependence of South Africa on landfill disposal, but further on the assumption that alternative treatment practices – such as recycling – are typically standardised procedures for which South Africa is likely to have good infrastructure, thus suggesting these practices should be well represented within existing LCA capacity. Furthermore, given the known limitation in LCA capacity for representing informal treatment practices — such as open burning and home composting — and the lack of representative data for these processes in the local context, interrogation of LCA capacity and adaptation potential for representing these processes is redundant.

3.3.3 Identification of Representative LCA Datasets

The review of datasets was limited to those contained within SimaPro v8.3 LCA software[5]. SimaPro is a commercial LCA modelling software, marketed as the "world's leading LCA software package", renowned for its transparency, cost effectiveness and reliability (PRé, 2017). While various software tools are available for LCA modelling, there are high demands on the software that it be flexible enough to model both standard scenarios as well as scenarios which diverge from the standard, while providing "ready-to-use" features and parametrisation such that the software is accessible to a wide variety of users (Unger, Beigl & Wassermann, 2004:2). According to Curran (2015:115), both GaBi and SimaPro

[5] SimaPro is a commercial LCA software developed by PRé Sustainability for application to both product and system LCAs. For more information, refer to https://simapro.com

are commercial LCA software packages capable of meeting these demands and have the "highest market share on a worldwide level". However, Curran (2015:115) further asserts that while process models in GaBi tend to rely on "aggregated system processes", SimaPro relies on "unit processes". The benefit of the unit process representation supported by SimaPro is that it provides greater transparency with regards to the system being modelled.

Given the objective of this research to have relevance to a product developer or LCA practitioner, SimaPro was considered a preferential basis for this investigation given its global application, capabilities in meeting the demands specified for a LCA software tool and unit process representation. A further benefit of assessing datasets within a commercially available LCA modelling software such as SimaPro is that it enables the interrogation of the applicability and adaptability of datasets on the platform through which this data is predominantly accessed.

An additional benefit of SimaPro is that it supports the ecoinvent database[6]. The ecoinvent database arguably provides the most comprehensive source of high quality, reliable and transparent LCI data (Frischknecht & Rebitzer, 2005). The ecoinvent database is frequently incorporated into dedicated LCA software and contains a centralised database containing LCI data on various systems such as energy, transport, waste treatment, chemicals and building materials (Frischknecht & Rebitzer, 2005). SimaPro v8.3 supports the latest ecoinvent release (v3.3) and hence can be considered a repository of current LCI data. Given that alternative sources of LCA data are available within SimaPro, the review of representative LCI datasets was not limited to those contained within ecoinvent v3.3 and considered all available datasets within SimaPro.

Open-source software, such as OpenLCA, also provides a prime example of LCA software. However, such software was not considered in this research. Although it is understood that OpenLCA is able to read ecoinvent data, it lacks capacity to support the modular construction of the ecoinvent database, thus providing a system view of the various processes inventoried as opposed to a unit process representation.

3.3.4 Assessment of Applicability of Datasets to South African Conditions

As detailed in Chapter 5, the only available datasets within SimaPro for the landfill disposal of general waste are those contained within the ecoinvent v3.3 database. Thus, the focus of this sub-section is directed towards the assessment of landfill disposal as represented within ecoinvent.

Following the identification of a representative landfill dataset, the dataset was reviewed with the objective of understanding the processes included in the inventory, the burdens associated with these processes and their representation within SimaPro. For ecoinvent v3.3 landfill datasets, the review process required the review of both the dataset itself and the underlying emission model used to generate the waste-specific burdens inventoried in the dataset. The underlying model is available as a downloadable Microsoft Excel file from the ecoinvent database (www.ecoinvent.org) for registered ecoinvent account holders[7]. Based on the findings from this review, each dataset was then assessed within the context of local conditions to ascertain their applicability to South African practices. An overview of this process and the associated objectives is shown schematically in Figure 3.3 overleaf.

[6] ecoinvent is a not-for profit organization providing a database for LCI process data. For more information refer to http://www.ecoinvent.org/home.html

[7] Both the 13_MSWIv2.xls and 13_MSWLFv2.xls files are necessary when undertaking dataset modifications. Waste composition data utilized in the landfill model (13_MSWLFv2.xls) is linked to the composition specified in the incineration model (13_MSWIv2.xls). Therefore, both spreadsheets are necessary if changes to waste composition are intended.

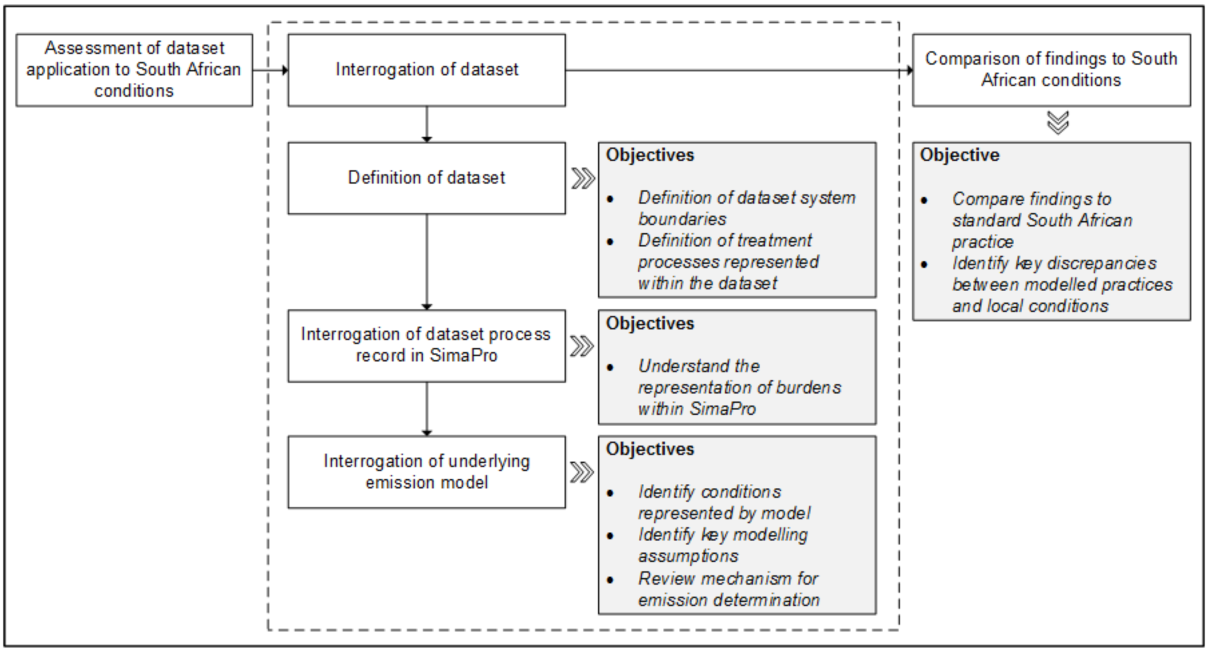

Figure 3.3 Methodological approach to interrogate the application of existing end-of-life datasets to South African conditions

3.3.5 Assessment of Dataset Adaptability

The final component of this stage of the research required an investigation into the parametrisation potential of existing datasets to facilitate improved representation of local conditions with regards to end-of-life LCA modelling. The parametrisation potential of both the dataset itself and the underlying emissions model was investigated, necessitated by the distinction between process- and waste-specific burdens within the ecoinvent landfill dataset. Process-specific burdens are independent of the waste that is landfilled and represent a specific infrastructure, resource or process-specific demand incurred during the landfill process. Waste-specific burdens by contrast represent the emissions associated with the degradation of the waste itself. The main or primary parameters considered for the adaptation of process- and waste-specific demands to local conditions are as follows:

- **Process-specific burdens:**
    - Process infrastructure
    - Energy demand
    - Material/resource demand
- **Waste-specific burdens:**
    - Leachate emissions
    - Landfill gas emissions

Each of the parameters shown above are in themselves dependent on various secondary parameters. Given the comprehensive nature of LCA inventories and the complexity of landfill modelling, an exhaustive assessment of parameters contributing towards these burdens is challenging. However, in alignment with the scope defined for this project, investigation was limited to those parameters considered to a.) have the potential to significantly influence landfill burdens and b.) be feasibly adjustable by a product-developer or LCA practitioner. According to Doka (2003d), waste-specific emissions from a landfill typically represent the most important burden within the LCIA result. Therefore, investigation of the parameterisation potential of waste-specific burdens was prioritised. This was, however, mostly limited to the generic modifications possible within the SimaPro process record.

3.4 Methodological Approach: Stage 3

The final stage of the research (key questions vi – vii) explored the extent to which modification of existing LCA datasets improves representation of local practices. This was undertaken by means of a case study approach. Four representative South African landfill scenarios were defined and the current ecoinvent landfill emission model and corresponding datasets modified to develop LCIs for these scenarios. Modifications were limited to those that could be considered feasible to be undertaken by a LCA practitioner or product developer for application within a product LCA. Although more detailed and complex modification is possible, this was not the focus of this work. The relative impacts of each scenario were then determined for different waste types by means of a LCIA approach.

The research approach was directly dependent on the findings from the previous stages and as such, the detailed methodology for this section is presented in Chapter 6, following the results from the previous stages. The following sub-sections are intended to provide an overview of the approach taken in terms of the definition of landfill scenarios, choice of materials and the LCIA approach.

3.4.1.1 Landfill Scenarios

The definition of representative landfill scenarios for the disposal of general waste in South Africa was informed by the status quo results presented in Chapter 4. An overview of the five different scenarios considered is shown below. Further detail on each scenario is available in Chapter 6, Section 6.1.

- **Scenario A:** Swiss sanitary landfill
    - Baseline landfill scenario
    - Inventory as represented within the current ecoinvent sanitary landfill dataset
- **Scenario B:** ‘Best case’ South African sanitary landfill
    - LFG capture efficiency = 75%
    - Agricultural application of wastewater treatment plant (WWTP) sludge
- **Scenario C:** ‘Average’ South African sanitary landfill
    - LFG capture efficiency = 0%
    - Agricultural application of WWTP sludge
- **Scenario D:** Non-engineered South African landfill
    - Daily site operations including compaction and covering
    - LFG and leachate capture = 0%
- **Scenario E:** Uncovered, open dump
    - No infrastructure or site operations
    - LFG and leachate capture = 0%

3.4.1.2 Waste Type

The performance of each scenario was assessed for three different materials. Given the focus of the work on general waste, three non-hazardous materials likely to be found in domestic waste were selected. Furthermore, given the temporal dependence of landfill emissions, it was of interest to compare the environmental impact of the landfill of materials with different organic content and degradability. This allows a comparison of the short- and long-term impact potential of each scenario for different waste types. The choice of material was intended to span the range of material likely to be found in general domestic waste as opposed to offering a statistical representation of waste types.

The following waste materials were selected:

- Polyethylene

    - Within the ecoinvent database, no distinction is made between low density and high density waste polyethylene. This material therefore represents a wide range of post-consumer products likely to be found in domestic waste, specifically plastic packaging. It is intended that this material represent slowly degrading waste (degradability defined within the ecoinvent dataset as 1%).
- Cardboard
    - Within the ecoinvent database, waste cardboard is inventoried as a separate material to paper. This material therefore represents a post-consumer packaging products with a high degradability (defined within the ecoinvent dataset as 32.4%).
- Compostable material
    - This category was in accordance with that defined within the ecoinvent database and reflects the composition of compostable material contained within average MSW derived by Hellweg (2000). This excludes paper, cardboard material and textiles. This material therefore represents the non-packaging component of domestic waste with a medium-high degradability (defined within the ecoinvent dataset as 27%).

Current ecoinvent landfill modelling capacity includes parameterisation of waste type. A specific waste type can be specified by the user within the model. Therefore, as the first step in developing an inventory for each scenario, the waste type was specified in the existing model structure (for further detail, refer to Appendix C, Section C.2). The elemental composition, waste degradability, upper and lower heating values and water content of each waste type considered are available in Table C.6 (Appendix C, Section C.3).

3.4.1.3 LCIA Approach

The LCIA of each landfill scenario was undertaken using the ReCiPe Midpoint (H) v1.13 method as available in SimaPro v8.3. This method was selected as it is the most recent method, with a wide coverage of impacts. The Europe Recipe H normalisation/weighting set was specified for the normalisation calculation. The midpoint method was selected as it focuses on single environmental problems. This not only allows the generation of results that are comparable to literature studies focusing on similar categories, but further has a lower uncertainty that the three highly aggregated levels used by endpoint indicators (i.e. effect on human health, biodiversity and resource scarcity) (Dutch National Institute for Public Health and the Environment, 2011). The Hierarchist version (H) of the method was selected as it is the default ReCiPe midpoint method and provides a moderate perspective between the Egalitarian (E) and Individualist (I) versions of the method.

Each scenario was assessed from both a short- (0 – 100 years) and long- (0 – 60 000 years) time perspective. Furthermore, the South African specific scenarios were modelled with and without leachate emissions. The following impact categories were modelled:

- Climate change
- Ozone depletion
- Terrestrial acidification
- Freshwater eutrophication
- Human toxicity
- Photochemical oxidant formation
- Particulate matter formation
- Terrestrial ecotoxicity
- Freshwater ecotoxicity

- Marine ecotoxicity

The LCIA results for the landfill of three different materials (polyethylene, cardboard and compostable material) were generated using SimaPro v8.3. Each dataset developed (see Chapter 6, Section 6.1 for detail regarding the development of each) was specified within SimaPro and evaluated using the ReCiPe Midpoint (H) v1.13 method and the Europe Recipe H normalisation/weighting set.

The output from SimaPro provided the normalised potential impacts for the desired categories (shown above), which were then presented in graphical format. The normalised potential impacts were expressed, as according to convention, as “Person Equivalents” (Pe). This unit provides an indication of the impact potential per person per year, that is to say, reflecting the fraction of the contribution to the impact deriving from the average person in the affected area. For consistency with standard reporting, results were presented as mPe (impact potential per person per year*1000). Using parameterisation within SimaPro, these impacts were assessed from both a short (0 – 100 years) and long (0 – 60 000 years) time horizon. It should be noted that the long-term time horizon is inclusive of short-term emissions.

3.5 Assumptions, Limitations and Uncertainty

The assumptions, limitations and uncertainties associated with this research and the implications thereof are discussed in depth where they arise in the results chapters. However, the following section provides an overview of the major assumptions, limitations and uncertainties associated with this research.

3.5.1 Assumptions

The major assumptions made in Stage 1 were related to the development of the model to provide a quantified estimate of informal waste flows. Key assumptions were linked to population distribution, income grouping and WGRs appropriate for these groupings. These assumptions are outlined and discussed in detail in Section 4.4.1.

With regards to the LCA and LCIA component of this research (Stage 3), it was assumed that the ReCiPe Midpoint (H) v1.13 method, as available in SimaPro v8.3, and the Europe Recipe H normalisation/weighting set specified for the normalisation calculation provides an indication of relative environmental impacts that can be interpreted within the South African context.

3.5.2 Limitations

The focus of this research was limited to general (municipal waste) with a specific focus on domestic waste. This focus was selected as it was intended that this research be of relevance to product stewards and LCA consultants evaluating end-of-life design for unequal societies where post-consumer waste management practices differ significantly. While waste will be generated industrially, such waste falls outside the scope defined for this research. This decision was based on the assertion that the majority of industrial waste is managed within the formal sector and hence does not have the inequality inherent in post-consumer waste management practices.

3.5.3 Uncertainties

The quantification of waste flows in South Africa (Chapter 4) was based on publicly available waste data from a variety of sources. Due to the nature of waste reporting and the limitations outlined by these sources with regards to the methodological approach utilised, this data is anticipated to reflect a high degree of uncertainty. However, no indication of the magnitude of this uncertainty was available, thus this data was utilised and presented with a general caveat regarding its uncertainty.

A second source of uncertainty lies in the ecoinvent datasets regarding the potential emission flows from the three waste types selected for the study. This in turn leads to uncertainty in the overall LCIA result. However, it was the intention of this work to compare the relative impacts associated with different waste management options as opposed to providing a quantified estimate and thus, analysis of these uncertainties was not considered relevant within the scope defined for this research.

3.6 Research Ethics

Research was conducted in accordance with the ethical requirements outlined by the Faculty of Engineering and Built Environment at the University of Cape Town (UCT). To ensure that the relevant ethical requirements were met, ethics clearance was obtained from the UCT Engineering and Built Environment Ethics in Research Committee. A copy of the ethics approval form is available in Appendix D. Although the nature of the research was regarded as low risk, due to the stakeholder engagement process, the following provisions were made to ensure compliance with the ethical requirements:

- The intended purpose and nature of the study was disclosed to the stakeholders in full prior to the formal engagement session.
- The confidentiality of the stakeholder was protected (if requested).
- Permission was requested from the stakeholder to record the interviews.
- The stakeholders were offered the chance to review the dissertation prior to its publication.

Chapter 4

STATUS QUO — WASTE MANAGEMENT IN SOUTH AFRICA

A critical finding from the literature review highlights the discrepancies in the level of waste service and management practises between developed and developing countries. Consequently, the application of LCA in representing product end-of-life in developing countries is limited, due in part to limitations in the availability of models and datasets capable of representing local waste management practices. As a first step in addressing this limitation and modelling the end-of-life stage of a product disposed of in South Africa, it is necessary to determine its fate within the local context.

This chapter provides a status quo analysis of general waste management in South Africa. It starts by presenting an overview of municipal waste management in South Africa, focusing on the legislative framework and regulatory requirements in addition to the suitability of existing data for mapping waste management. This is followed by an overview of informal domestic waste management. Thereafter, an investigation into landfill disposal practices undertaken in South Africa is presented. This investigation identifies characteristic landfill practices, provides a basic mapping of waste flows between different landfill sites, and assesses the compliance of these sites to regulatory conditions. The final section of this chapter presents a synthesis of these findings, presenting a possible representation of the fate of general waste in South Africa.

4.1 Overview of Municipal Waste Management in South Africa

According to global standards, South Africa is classified as a developing country. However, in some respects, this classification is contentious. As the biggest economy of the Southern African region (World Bank, 2018), South Africa reflects a number of the characteristics of a developed country with a high level of urbanisation, wealth, and infrastructure. However, this level of development is unequally distributed amongst the population, and a notable disparity exists from both an economic and social perspective. The effect of this disparity is particularly evident in service delivery; according to the *Community Survey 2016* (Stats SA, 2016b), households in rural municipalities typically receive fewer and inferior services to those households in more affluent, urban municipalities (Stats SA, 2016c). Whilst various factors contribute towards this discrepancy, significant factors include the historical inequalities in development (evident predominantly in former homeland areas), the high level of poverty (which is prohibitive to households' ability to pay for services), and the practical and financial constraints associated with extending services to remote rural or inaccessible informal areas (Stats SA, 2016c).

Waste management in particular illustrates this discrepancy. Within South Africa, waste management is the responsibility of local government, however, large discrepancies exist in the level of service provided by different municipalities. Where most larger municipalities provide a complete service, including waste collection and appropriate disposal (albeit not consistently meeting regulated environmental controls), many smaller municipalities — typically rural — lack the capacity for any form of waste service delivery (Friedrich & Trois, 2010). While the range in waste service delivery is most notable between urban and rural municipalities, it has been suggested that variation can, and does, occur across provinces, district councils, and local municipalities (Stats SA, 2016c). For example, while relatively affluent urban areas typically receive a complete waste service, certain urban communities —

specifically in informal settlements — lack even basic refuse removal (Stats SA, 2016a). This can in part be attributed to the fundamental, unserviceable nature of informal settlements, where limited road access, high settlement density, poor spatial planning and layout of settlements, and illegal land tenure prohibit the delivery of waste collection services (von der Heyde, 2014).

Given the disparity in the provision of formal waste management services across the country, the determination of a representative waste scenario for South Africa is complicated, requiring consideration of both formal and informal management scenarios. According to the most recent national household survey (Stats SA, 2016a), 31% of households do not receive a regular waste collection service. The lack of formalised waste management can result in informal management practices such as illegal dumping, uncontrolled burning, the operation of illegal dumpsites, and unlicensed landfills (Fiehn & Ball, 2005). Such practices typically provide poor management and containment of the waste stream, resulting in a high potential for leakage of waste and contaminants into the environment (Fiehn & Ball, 2005). The distinction between formal and informal waste management practices based on waste service delivery does not necessarily provide an indication of “good” and “bad” environmental performance. For example, while the environmental impacts of landfilling can be controlled if operations are adequately managed, so called formal landfill sites can vary in terms of design and management, with shortcomings in siting, design and operation presenting a significant potential for pollution (Department of Water Affairs and Forestry [DWAF], 1998c).

While the definition of waste management scenarios based on the provision of waste service delivery is convenient, the variation in the licence status of waste disposal facilities challenges the definition of what can be constituted as formal disposal. It is a requirement laid out in the *National Environmental Management: Waste Act, No. 59 of 2008* (2009:chap 5) (hereafter, *NEM:WA*) that all waste management activities with the potential to cause a detrimental effect on the environment obtain a waste management license[8]. Although the current proportion of unlicensed waste management facilities is unknown, in 2006, according to the Department of Environmental Affairs and Tourism (DEAT) (2006) 760 sites (both illegal and legal) were operating without a permit. This figure excludes small, unrecorded sites in rural areas, and hence with their inclusion, this figure is likely to be higher. This implies that waste that is formally collected can be disposed of in either municipal or privately owned and managed, but unlicensed landfill sites or treatment facilities. Furthermore, waste which is informally disposed of can be recovered and enter into the formal waste stream. This is typically seen in the municipal clearing of dumped waste, or the informal scavenging of recyclable materials from dumped or littered waste.

Given the potentially adverse social and environmental consequences of informal waste management practices, characterisation and quantification thereof can be beneficial in assessing the magnitude of the problem and identifying mitigation actions. Characterisation and quantification of informal practices presents an obvious challenge in terms of data availability. The challenge of data availability is not however limited to informal practices, and even for so called formal waste management there is a paucity of consistent and reliable waste data (Department of Environmental Affairs [DEA], 2012a).

4.2 Application and Uptake of Waste Legislation in South Africa

Waste management in South Africa is governed by various pieces of legislation. Relevant legislation as identified on the South African Waste Information Centre (SAWIC) are listed in Table A.1 (Appendix A). Historically, waste legislation in South Africa had been fragmented, the effect of which can still, to an

[8] Waste management activities requiring a license in terms of Section 19(2) of the *NEM:WA* (2009) are available in the *National Environmental Management Waste Act, No 59 of 2008. List of Waste Management Activities that have or are likely to have a Detrimental Effect on the Environment* (2013).

extent, be observed in the current legislative framework (DEA, 2011a). This historical limitation has in the most part been overcome with the promulgation of the *NEM:WA* (2009), which is the centrepiece of waste legislation in South Africa, providing the basis for the development and implementation of supporting legislation, policies, and regulations. Further detail and discussion on key pieces of waste legislation, starting with the *NEM:WA* (2009), and followed by supporting policies and regulations outlining the responsibilities of national, provincial, and municipal governments in implementing and enforcing this legislation, is available in Appendix A, Section A.1.

Since its promulgation, the *NEM:WA* (2009) has been hailed for is alignment with global best practice for waste management, enshrining such principles as “prevention”, “the polluter pays”, and “separation at source”, which in essence denotes a departure from existing waste management structures typically based on end-of-pipe disposal (Coetzee, von Blottnitz & Hamann, 2014:264). Despite providing world-class legislation, adherence to this legislation is lacking. For example, 31% of households are un-serviced in terms of waste services (Stats SA, 2016a), indicating that a number of municipalities are not meeting their obligations for waste collection. Furthermore, despite being a legislated requirement in terms of Section 4(4.1) of the *National Environmental Management: Waste Act, No 59 of 2008. National Domestic Waste Collection Standards* (2011), separation-at-source programs are not currently in operation in all areas of metropolitan and secondary cities, again pointing to the overall gap between the aspirational aspect of the *NEM:WA* (2009) and the reality on the ground.

Given its departure from traditional waste management systems such as landfill disposal, it has been suggested that the implementation of the *NEM:WA* (2009) is complex, requiring changes to both existing systems and infrastructure, which has far reaching implications in terms of capital funding and status quo changes (Coetzee, von Blottnitz & Hamann, 2014). Section 7(2) of the *NEM:WA* (2009) makes provision for the regionalisation of waste management services, with the role of the municipality defined in Section 9. However, the implementation of alternative waste management strategies at a municipal level has been largely “ad hoc”, constrained by limitations in municipal capacity, and the high costs associated with alternative disposal (Sango et al., 2014:225). Furthermore, historical imbalances in access to waste management services in South Africa has resulted in a backlog of waste services in predominantly urban informal areas, tribal areas, and rural formal areas (Zhakata et al., 2016). Under such conditions, the provision of a basic service represents a key priority to municipalities. This stands in contrast to areas in which a waste service is well established, allowing municipalities to focus on meeting the new waste management objectives and prioritise improvements to the current system in alignment with the principles of the waste hierarchy.

Devolving responsibility for the actual implementation of waste services to local government only works where there is a functioning local government, however, this is not the case in many South African municipalities. While municipal ineffectiveness in South Africa is partially due to shortcomings in the performance of the municipality itself (i.e. inadequate service delivery and a culture of “self-enrichment on the part of municipal councillors and staff” (Atkinson, 2007:53)), poor municipal performance is compounded by the failure of the intergovernmental system to provide adequate support to local government (Atkinson, 2007). As provided for in Section 9(4) of the *NEM:WA* (2009), where a municipality acts in conflict with either national or provincial waste management standards and fails to meet its waste management obligations, the municipality should receive intervention and support from provincial and national government spheres (Odeku, 2014). However, the failure of organs of state in adhering to the stipulated corporative governance is common (Zhakata et al., 2016).

4.3 Waste Quantification and Characterisation in South Africa

The implementation of, and adherence to, the principles of the *NEM:WA* (2009) are challenged by a number of factors. Thus, the reality of waste management in South Africa reflects a departure from that which is legislated. This section is directed towards establishing the status quo with regards to waste management in South Africa. Limitations in the availability of consistent and accurate waste data have been recognised as a challenge in this regard. This section therefore simultaneously aims to interrogate the state of knowledge and availability of representative data within this sector.

4.3.1 South African Waste Information System

The South African Waste Information System (SAWIS) is the national waste information system (WIS) established in terms of Section 60 of the *NEM:WA* (2009). It is intended that the SAWIS address the paucity of reliable waste data in the country by providing a national repository of waste data for both organs of state and the public, containing information on the quantity and type/classification of waste generated and managed in addition to providing a register of licensed waste management activities. It is further intended that the SAWIS provide information on the level and extent of waste management services provided by municipalities and other information regarding compliance with national waste regulations and standards (*NEM:WA*, 2009:s60, ss2). The framework for the SAWIS system in terms of data providers and reporting structure is shown in Figure 4.1.

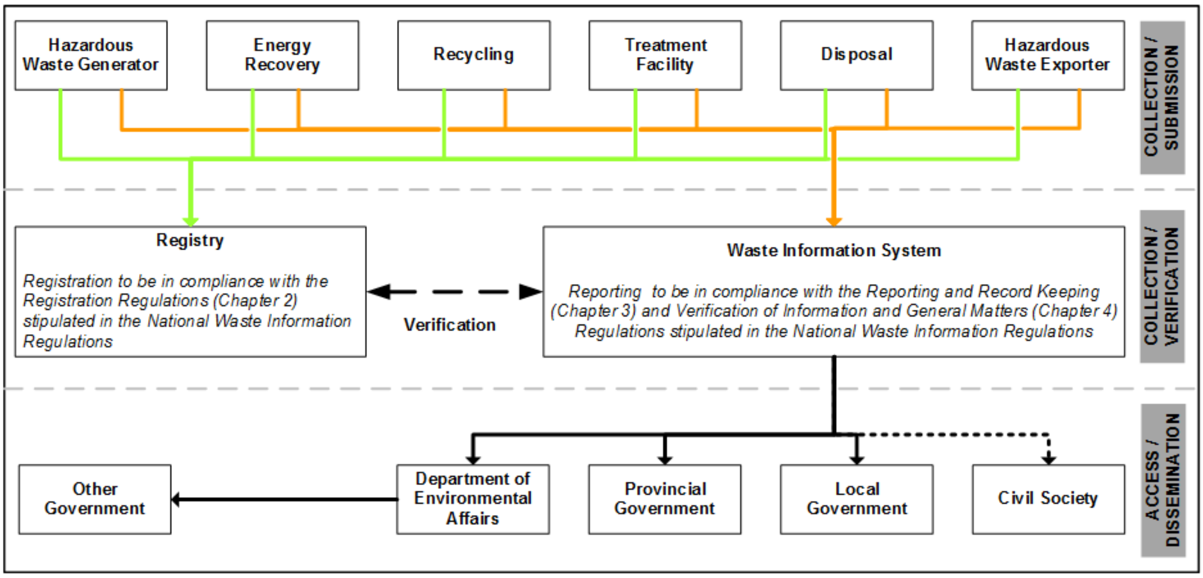

Figure 4.1 Framework showing the data providers and reporting structure to the SAWIS (Adapted from DEAT, 2005:5)

The SAWIS system was developed and piloted by the DEAT between 2004 and 2006 as part of the *National Waste Management Strategy Implementation Project* (DEA, 2012a, DEAT, 2005). Since the completion of the project, reporting to SAWIS continued, albeit on a voluntary basis, thus limiting the functionality of the system in representing the country as a whole (DEA, 2012a). It was anticipated that this shortcoming would be overcome following the promulgation of the *National Environmental Management: Waste Act, No. 59 of 2008. National Waste Information Regulations* (2012), developed with the objective of regulating the collection and reporting of waste data to the SAWIS, as per Section 60 of the NEM:WA (2009).

The regulated reporting requirements suggest that the SAWIS should provide a comprehensive source of waste data for South Africa. Public access to the waste data reported to the SAWIS is available online through the SAWIC (DEA, 2017b). An overview of general waste tonnages reported to the SAWIS for

2016, presented by type of waste activity and management option for the country as a whole, is shown in Figure 4.2. An explanation of the relevant waste management codes is available in Table 4.1.

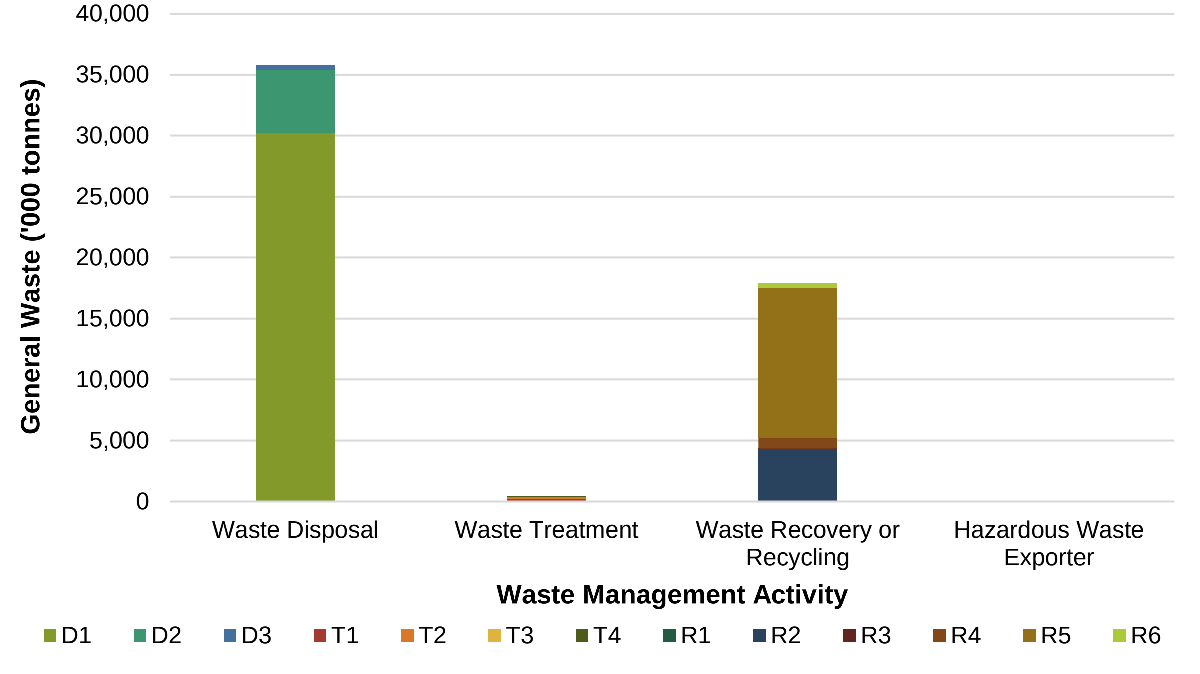

Figure 4.2 2016 SAWIS data for general waste tonnages by waste management activity and management option (data generated using SAWIC tonnage reports last updated 20 July 2017 (DEA, 2017b)). An explanation of the relevant waste management codes is available in Table 4.1.

Table 4.1 List of waste management options and corresponding disposal, recycling, recovery and treatment codes as defined in Annexure 5 of National Environmental Management: Waste Act, No. 59 of 2008. National Waste Information Regulations (2012)

	Code	**Economic Instrument**
Disposal	**D1**	Disposal of waste to land (e.g. specially engineered landfill)
	D2	Disposal of waste to landfill (e.g. non-engineered landfill)
	D3	Storage/disposal of waste in surface impoundments (e.g. placement of liquid or sludge discards into pits, ponds, lagoons etc.)
	D4	Release of waste into water bodies (except seas/oceans)
	D5	Permanent storage (stabilization, micro-encapsulation, macro-encapsulation)
Treat-ment[a]	**T1**	Biological treatment (e.g. biodegradation, composting, biogas generation)
	T2	Physical treatment
	T3	Chemical treatment
	T4	Thermal treatment (incineration, pyrolysis etc.)
Recycling and Recovery	**R1**	Direct recovery of energy from waste
	R2	Direct recovery of raw material from waste
	R3	Regeneration or rejuvenation of waste (solvents, carbons, acids and alkalis)
	R4	Recycling of organic substances
	R5	Recycling of metals and metal compounds
	R6	Recycling of other inorganic materials

[a] Treatment not for the purposes of disposal

According to Figure 4.2, in 2016 waste management operators reported a total of 54 million tonnes of general waste. The breakdown of this figure in terms of the specific waste types as categorised in Annexure 2 of the *National Environmental Management: Waste Act, No. 59 of 2008. National Waste*

Information Regulations (2012) is available in Table A.5 (Appendix A.2). The validity of Figure 4.2 in representing the total quantity of general waste generated and managed in South Africa is contentious. Firstly, the total quantity of general waste reported under each management option is based on the sum of the individual waste categories reported in Table A.5. The problem with this approach is that the waste categorisation used — although compliant with national regulations — introduces the potential for double counting, as many of the general waste categories represent waste types that are also reported under separate categories. For example, recyclable material such as paper (GW50), plastic (GW51), and glass (GW52) can be contained in both municipal (GW01) and commercial and industrial waste (GW10), and it is unclear to which extent, if any, reporting facilities report on the individual material fractions of a mixed stream such as GW01 or GW10. Secondly, certain management options (i.e. D1 and R4) are recorded under dual waste management activities. Thus, it is uncertain whether in the determination of total waste managed, this constitutes double counting.

Figure 4.2 further suggests that in 2016, approximately 67% of general waste was disposed of, 32% was recovered for recycling or energy, and less than 1% of general waste underwent treatment. Of the disposal fraction, 84% of the waste was reported under D1 (specially engineered landfills, 114 reporting facilities) and 15% under D2 (non-engineered landfills, 133 reporting facilities). Despite the regulatory requirements for reporting to the SAWIS, compliance remains constrained, with only a proportion of active waste management facilities reporting data to the system and, furthermore, doing so with a questionable accuracy (Hanekom, personal communication 2016, 30 May 2016).

While the number of facilities reporting to the SAWIS has the potential to affect the waste tonnages reported by the system, the quality of the reporting should also be considered. With regards to the latter, analysis of Table A.5 shows that the level to which facilities report data is inconsistent, with some facilities reporting waste to Level 3 (specific waste type) and others reporting a total quantity under Level 2 (major waste type). This imposes further potential for double counting, as it is unclear as to whether data that is reported to a Level 2 category is additionally disaggregated under Level 3. Furthermore, a review of the SAWIS data undertaken by the DEA (2012a) found the system to contain a number of inaccuracies, such as order of magnitude changes in the reported quantities of waste from month to month, suggesting errors in the input of data (i.e. incorrect placing of the decimal point). Additionally, order of magnitude differences in the quantity of waste reported by facilities of similar sizes was observed. It is unclear whether data capturing improvements have been made since. Further discussion and examples of the current limitations in the SAWIS as a source of representative waste data are available in Appendix A, Section A.3.

Despite current limitations in reporting to the SAWIS, the potential value of a mandated national WIS is significant. The functionality of the Western Cape Integrated Pollutant and Waste Information System, for example, provides an illustration of the benefits of a comprehensive and fully-functioning WIS. This system was established as per Section 62 of the NEM:WA (2009) and is regarded as a comprehensive source of provincial waste data, used extensively in the development of provincial and municipal Integrated Waste Management Plans (IWMPs), and other waste planning. Maintaining the functionality of this system is supported by means of directed education, and stringent data verification and auditing processes (Hanekom, personal communication 2016, 30 May 2016). Therefore, if a similar dedicated approach is undertaken with regards to the SAWIS, it is anticipated that with time, both the standard of reporting to the system and the coverage of waste management facilities within the system will continue to improve.

A limitation to the level of representation that SAWIS can provide lies with the continued operation of informal or unregulated waste management operations. Considering the results of the *General*

Household Survey 2015 (Stats SA, 2016a): 28% of South African households reportedly dispose of waste in their own refuse dumps, and in rural areas this is as high as 81%. Although private disposal sites are unlikely to meet the daily waste deposition volumes necessary to obtain a waste license (and hence report data to the SAWIS), assuming that these sites are unlikely to have a high level of engineering control, counting this waste in addition to that reported into the SAWIS system would increase both the total quantity and relative proportion of waste disposed of under D2. Therefore, so long as informal and unregulated sites continue to manage and treat waste, the waste volumes reported to the SAWIS will under-represent waste volumes in the country. While this limitation can be addressed to a certain extent with the enforcement of waste legislation (i.e. increasing licensing of facilities and closing facilities where necessary), before such an approach can be effective it is necessary to address the disparity in waste service delivery — particularly in order to eliminate the dependence of un-serviced households on private dumps and other alternative waste management practices.

Even though reporting to the SAWIS is a license condition for waste management facilities, increasing the licensing of sites will not guarantee that licensed operations will report data. Additionally, in lieu of appropriate infrastructure (i.e. functioning weighbridges) and adequate training, the quality of data reported to the system is likely to remain constrained (Käsner, personal communication 2016, 15 May 2016). Therefore, although increased licensing will enable a more comprehensive mapping of waste management facilities operating in South Africa, this will not necessarily improve waste data per se or facilitate a more robust analysis of waste management as a whole. To enable more robust analysis, information is needed with regards to compliance to these permit conditions in terms of reporting and the accuracy thereof (Godfrey, 2008).

4.3.2 The National Waste Baseline

In lieu of a fully representative and verified national waste repository, arguably the most comprehensive source of national waste data is contained within the *National Waste Baseline Report* (DEA, 2012a) (hereafter, *NWBR*). This report was developed to provide a national baseline against which the implementation of the *NEM:WA* (2009) and subsequent *National Environmental Management: Waste Act, No. 59 of 2008. National Waste Management Strategy* (2012) (hereafter, *NWMS*) could be tracked (DEA, 2012a). According to the *NWBR* (DEA, 2012a), in 2011 South Africa generated approximately 108 million tonnes of waste, of which ±90% was disposed of in landfills, with the balance recovered for recycling. However, given the acknowledged limitations in the study, according to von Blottnitz (2016), this figure could be better reported as 110 ± 20 Mt (where 1 Mt = a million metric tonnes).

The waste categories reported in the *NWBR* (DEA, 2012a) are in accordance with the *National Environmental Management: Waste Act, No. 59 of 2008. National Waste Information Regulations* (2012:s8). Of the total waste, 55% was classified as general waste, 1% as hazardous waste, with the balance reported as unclassified waste. Although the Regulations do not specify a separate category for uncategorised waste, certain waste types appear under both general and hazardous categories. According to the *Minimum Requirements for the Handling, Classification and Disposal of Hazardous Waste* (DWAF, 1998a), dual categorisation is necessary as certain waste types, although presenting a relatively low hazard based on their intrinsic properties, are potentially hazardous due to their high volumes. The level to which waste data was available in the development of the *NWBR* (DEA, 2012a) was recognised as prohibitive in distinguishing between the general and hazardous component of certain waste streams. The unclassified waste category was therefore introduced so as to not skew the results of the baseline (DEA, 2012a). A summary of disposal data for various general waste streams as reported in the *NWBR* (DEA, 2012a) is shown in Table 4.2 overleaf.

Table 4.2 Overview of general waste streams by management option for South Africa in 2011 (Adapted from DEA, 2012a:15)

Code	General Waste Categories	General Waste by Management Option			
		Generated	Recycled	Landfilled	Recycled
		Kilo tonnes			%
GW01	Municipal waste (non-recyclable portion)[a]	8063	-	8063	0
GW10	Commercial and industrial waste[b]	4233	3259	973.5	77
GW30	Construction and demolition waste	4 726	756	3 969	16
GW20	Organic waste	3024	1058	1965	35
GW50	Paper	1734	988.6	745.7	57
GW51	Plastic	1309	235.5	1073	18
GW52	Glass	959.8	307.1	6527	32
GW53	Metals	3121	2497	624.2	80
GW54	Tyres	247	9.865	237	4
GW99	Other	36 171	-	36 171	0

[a] In order to avoid double counting, only the non-recyclable portion is represented here. Recyclable fractions are represented separately.
[b] According to DEA (2012a), commercial and industrial waste represents 21% of total municipal waste with the recyclable content accounted for under separate categories. It therefore follows that the summation of general waste categories to determine the total general waste must exclude commercial and industrial waste.

According to Table 4.2, in 2011 South Africa generated a total of 59 million tonnes of general waste — though the bulk (36 Mt) of this is classified as "other", representing a time-inflated estimate of an earlier uncertain estimate (von Blottnitz, 2016). Given that waste generation is dependent on population, extrapolating the 2011 general waste quantity to 2016 using a per capita general waste generation rate (determined from the 2011 population statistics reported in the *NWBR* (DEA, 2012a:8)) suggests that the quantity of general waste generated in South Africa in 2016 will be in the order of 63 million tonnes. Comparison of this figure to the 54 million tonnes of general waste reported to the SAWIS for 2016 (see Section 4.3.1) appears to support the assertion that the SAWIS provides a limited representation of the waste sector. The comparison of these quantities must be undertaken with caution, particularly given the high uncertainty range associated with the *NWBR* (DEA, 2012a) results suggested by von Blottnitz (2016) and the waste characterisation utilised by each source.

The latter presents a particular constraint to the direct comparison of results due to the "unclassified waste" category defined by the *NWBR* (DEA, 2012a:5). As a result of this classification, the results shown in Table 4.2 exclude a number of general waste categories defined under Section 8 of the *National Environmental Management: Waste Act, No. 59 of 2008. National Waste Information Regulations* (2012). The tonnages reported to the SAWIS, by contrast, do not make such a distinction. Therefore, for the sake of consistency, the tonnages of waste defined as unclassified in the *NWBR* (DEA, 2012a) should be deducted from the SAWIS tonnages before comparing the two sources. According to the *NWBR* (DEA, 2012a:18) unclassified general wastes include brine (GW13), fly ash and filter dust (GW14), bottom ash (GW15), slag (GW16), mineral waste (GW17), e-waste (GW18), and sewerage sludge (GW21). Deducting the corresponding tonnages reported to the SAWIS for each of these categories (using the results available in Table A.5) yields a revised total of 34 million tonnes of general waste reported by SAWIS. The discrepancy between this result and that suggested by the *NWBR* (DEA, 2012a) is significant, further suggesting that either the tonnages reported to SAWIS under-represent the total general waste generated in the country or supporting the notion that the GW99 ("other") figure in the *NWBR* (DEA, 2012a) is a phantom number, which highlights the ongoing lack of reliable and comparable waste data in South Africa.

Analysis of Table 4.2 further suggests that South Africa has a relatively high recycling rate for a number of general wastes (i.e. GW10, GW20, GW50, GW52, GW53), which according to von Blottnitz (2016), bears testimony to the advanced state of recycling in the country. Notable exceptions are both plastics (GW51) at 18% and tyres (GW54) at 4%. Although the high recycling rates suggested by the *NWBR* (DEA, 2012a) for certain wastes are encouraging, according to the *NWBR* (DEA, 2012a), landfill represents the only alternative to recycling. Although the *NWBR* (DEA, 2012a) represents the status quo with regards to waste management post the promulgation of the *NEM:WA* (2009), it precedes the *NWMS* (2012). This provides a possible explanation for the apparent lack of alternative treatment options for waste in South Africa.

Comparison of Table 4.2 with the SAWIS data in Figure 4.2 would suggest that limited changes have occurred to the state of alternative waste management options in the country since the publication of the *NWBR* (DEA, 2012a). Analysis of the SAWIS data suggests that although "treatment" and "waste recovery and recycling" are presented as alternative options to landfill disposal for the majority of general waste categories, the tonnages of waste reported under "treatment" are very low. The limited uptake of alternative treatment options has been attributed to a number of challenges, primarily financial in nature. For example, although well established in many countries, waste-to-energy schemes such as incineration with energy recovery have not yet been implemented in South Africa due to their prohibitive costs compared to landfill disposal (von Blottnitz, 2016). A comparison of the tonnages of waste reported to the SAWIS and by the *NWBR* (DEA, 2012a) for the management of the non-landfilled portion of general waste is available in Table A.7 (Appendix A.3).

Given the variation in the level of waste service provision in South Africa, it is uncertain to what extent informal waste management is incorporated into the results shown in Table 4.2. The under-representation of waste quantities and management options is recognised as an important limitation to the *NWBR* (DEA, 2012a) results. It is explicitly stated in the document that that the findings presented are exclusively focused on "waste in official waste streams and are therefore likely to be an under estimate of the total waste generated and disposed of in South Africa" (DEA, 2012a:5). To illustrate this shortcoming, if it is assumed that that all waste generated in un-serviced households is not captured in formal estimates, then the *NWBR* (DEA, 2012a) has the potential to under-report the domestic proportion of general waste by 31% (based on the proportion of un-serviced households reported in the *General Household Survey 2015* (Stats SA, 2016a)). When this volume is considered in conjunction with other "unofficial" waste streams, such as illegal dumping and unrecovered litter, the proportion of unofficial waste could become significant. However, the *NWBR* (DEA, 2012a) is not explicit with regards to the extent such streams are incorporated into "official" waste streams, and in lieu of detailed accounting of dumped and littered waste in South Africa, the estimation of these flows is a challenge.

A lack of consistent and accurate raw data — even for so called 'official' waste streams — presents a limitation to the accuracy of the results of the *NWBR* (DEA, 2012a). The development of the national baseline was dependent on municipal waste data, noted for its variability in both accuracy and availability (DEA, 2012a). Municipal waste data is typically collected at landfills. Consequently, only waste disposed of in this manner is being accounted for, and is likely being done with questionable accuracy due to limitations in landfill infrastructure such as weighbridges (DEA, 2012a). According to DEA (2012a:7), the accuracy and comparability of available waste data is constrained by the following:

- Limited standardisation in sampling and sorting methods
- Variation in waste categorisation
- Unrepresentative sample sizes
- No consideration of season variation in sampling methods
- Variability in sorting accuracy

Although the development of the SAWIS was intended to address shortcomings in waste generation and characterisation data, at the time of publication of the *NWBR* (DEA, 2012a), reporting to SAWIS was voluntary, and thus the system was considered incapable of providing accurate waste data (DEA, 2012a). Analysis of the SAWIS data presented in Section 4.3.1 suggests that the use of this system remains constrained. In lieu of comprehensive reporting to SAWIS, the *NWBR* (DEA, 2012a) arguably remains the most representative source of national waste data in South Africa.

4.3.3 Waste Generation Rates

In the absence of a fully functioning and representative national waste data repository, per capita waste generation rates (WGRs) provide a cost effective and quick estimate of waste quantities without undertaking primary data collection (DEA, 2012a). Although this method provides an indication of total waste generation rather than the management distribution thereof, it is a useful starting point for determining waste generation volumes and assessing the recovery potential for different waste types. As noted in the literature review (Chapter 2), WGRs are linked to a number of factors including the population size, economic productivity, and the population distribution in terms of urban and rural settlement (Bogner et al., 2008). However, limited independent waste generation studies have been undertaken in South Africa, and of those that exist, limited socio-economic distinction in the WGR has been made (Myers & Pieterse, 2014). A comparison of different WGRs available for South Africa are shown in Table 4.3 overleaf.

The results shown in Table 4.3 indicate that the availability of WGRs representing South Africa is varied, with different sources providing a different level of disaggregation in terms of regional and socio-economic WGRs. The WGR itself varies in its definition, with some sources reporting a municipal WGR, some a domestic WGR, and others an unspecified WGR. While it is unlikely that Table 4.3 provides an exhaustive summary of available WGRs for South Africa, it provides further indication of the disparate and inconsistent nature of waste reporting in the country, and beyond.

Despite being based on 1998 waste generation and population statistics, the domestic WGRs presented by the DEAT (2006) (as shown in Table 4.3) appear to provide an official national WGR and are recommended for use in the determination of municipal waste quantities in the *Guideline for the Development of Integrated Waste Management Plans* (DEA, n.d.). Given that the socio-economic groupings used were not explicitly defined, this raises concern over how to conduct a reasonable grouping of individual or household income in South Africa, due to the high income inequality that exists (Myers & Pieterse, 2014). The Gini index is used by the World Bank to give an indication of income distribution within a country, and South Africa has one of the highest Gini indices in the world, reported at 0.7 in 2009 (Bosch et al., 2010) (where an index of 1 indicates complete inequality). The question of how to develop income groupings and establish a representative South African middle class has been explored by Visagie (2013), who recognises the complication that the low average and median levels of income, and wide income distribution have on this task.

As noted above, the definition of the WGR varies between sources, challenging the translation of this information into comparable national waste estimates. It is unclear whether domestic waste is comparable to municipal waste, and indeed to what extent municipal waste is representative of all general waste categories. Furthermore, extrapolating the domestic or municipal component of waste determined using WGRs to a national general waste quantity is dependent on accurate waste characterisation data. Characterisation studies are limited at both municipal and national level (Oelofse, Muswema & Koen, 2016), which makes accurate extrapolation challenging. Further detail on South African waste characterisation is available in Appendix A, Section A.4.

Table 4.3 Per capita waste generation rates for municipal and domestic waste

Source	Region	Domestic Waste Generation (kg/capita/day)				
		Low Income	Middle Income	High Income		
DEAT (2006)[a]	South Africa	0.41	0.74	1.29		
Department of Agriculture Conservation and Environment North West Provincial Government (2008)[b]	South Africa	0.2 – 0.7	0.7 – 1.9	1.5 – 3.0		
Gauteng Department of Agriculture and Rural Development (2009)	Johannesburg	0.38	0.66	0.99		
Ogola, Chimuka and Tshivhase (2011)[c]	Limpopo	0.32	0.4	0.7		
		General Waste Generation (kg/capita/day)[e]				
Fiehn and Ball (2005)[d]	Eastern Cape	0.31				
	Free State	0.55				
	Gauteng	2.08				
	Kwazulu Natal	0.43				
	Mpumalanga	1.42				
	North West	0.19				
	Northern Cape	1.50				
	Limpopo	0.28				
	Western Cape	1.85				
Gauteng Department of Agriculture and Rural Development (2009)	Sedibeng	0.2				
	West Rand	0.31				
	Tshwane	2.99				
	Gauteng	1.32				
		Metro	Large Urban	Medium Urban	Small Urban	Rural
Western Cape Government Department of Environmental Affairs and Development Planning (2010)[f]	Western Cape	2.50	1.50	1.20	0.75	0.10 – 0.57
		Municipal Waste Generation (kg/capita/day)				
IPCC (2006a)	Africa	0.79				
Hoornweg and Bhada-Tata (2012)	Africa	0.65				
Hoornweg and Bhada-Tata (2012)	South Africa	2.0				
Hoornweg and Bhada-Tata (2012)	Upper middle-income countries	1.16				

[a] Socio-economic groups were not defined in terms of income characterisation. Original source of data referenced to *Waste Generation in South Africa: Baseline Studies* by DWAF (1998) however this document was not available.

[b] Socio-economic groups were not defined in terms of income characterisation. This source quotes figures from IWMP Guidelines (DEA, n.d.), however, these figures in the original document could not be found.

[c] Socio-economic groups defined according to residential stand size: low income < 300 m^2, middle income 300 – 500 m^2, high income > 500 m^2.

[d] Original source of data referenced to *Waste Generation in South Africa: Baseline Studies* by DWAF (1998) however this document was not available.

[e] General waste defined as waste generated from households, commerce, institutions and the manufacturing industry.

[f] As quoted by Myers and Pieterse (2014:9). Original document could not be sourced online. No specification of the waste generation rate available.

4.3.4 Integrated Waste Management Plans

South African national government priorities regarding circular economy, stricter landfill standards, and shortages of landfill airspace have placed increasing pressure on municipalities to divert waste from landfills (Oelofse, Muswema & Koen, 2016). One mechanism to achieve this objective is that certain organs of state prepare Integrated Waste Management Plans (IWMPs), as laid out in the *NEM:WA* (2008:chap3, s11). The objective of the IWMPs, according to Goal 5 of the *NWMS* (2009:32), is to "align and integrate the actions of national, provincial and local government" by setting targets for waste management and "give practical effect to the policies and instruments set out in [the] NWMS". Goal 5 further stipulates that as the primary providers of waste management services, municipalities are required to develop effective and implementable IWMPs, which are in alignment with their Integrated Development Plans (IDPs).

It is stipulated in Section 12 of the *NEM:WA* (2009) that IWMPs must include, amongst other requirements —

- A situational analysis including an assessment of the quantities and types of waste generated and a description of the waste management services that are available
- Indication of intent to give effect to the objectives of the *NEM:WA* (2009) and give effect to best environmental practice in terms of waste management
- The establishment of targets for the collection, minimisation, re-use and recovery of waste

The framework and requirements for IWMPs are developed further in the *NWMS* (2012). This strategy encourages industrial sectors generating priority waste streams to develop and implement Industry Waste Management Plans (IndWMPs) to effectively manage waste through increased recycling and recovery measures (*NEM:WA*, 2008:s28). According to the *NEM:WA* (2008:s30, ss2), the contents of the IndWMPs are to be specified by the Minister or MEC and could include, amongst other aspects —

- The quantity of waste generated
- Intended measures to prevent pollution or ecological degradation
- Targets for waste minimisation through waste reduction, re-use, recycling and recovery
- Strategies to minimise waste generation and disposal
- Measures to be taken to manage waste
- Methods for monitoring and reporting

Following the requirements laid out for both IWMPs and IndWMPs, these documents therefore have the potential to provide a valuable source of information for establishing the status quo with regards to waste management in South Africa. However, according to the *NWMS* (2009:s1, ss1.6), one of the main challenges facing waste planning in South Africa is the unreliable and contradictory nature of waste data. The usefulness of IWMPs and IndWMPs in representing the status quo is further constrained by the nature of the plans themselves, which vary between municipalities and provinces. Thus, while a comprehensive, detailed plan provides a snapshot of waste management within one area of South Africa, this cannot necessarily be assumed to be representative of the country as a whole. Furthermore, extracting relevant information from a number of IWMPs to construct a broader picture of waste management is challenged by inconsistencies in reporting standards and methodologies. An analysis of publicly available IWMPs identified the following key constraints with regards to their use in developing a representation of waste management in South Africa:

- **Inconsistent reporting:** Although the content requirements of the IWMPs are outlined in Section 12 of the NEM:WA (2009), these are relatively broad, resulting in inconsistent categorisation of waste and reporting classes. This limits both the comparability of data, and the aggregation and extrapolation of data to represent a national scenario.

- **Methodology:** Due to a lack of standardised methodology for waste characterisation, various approaches were undertaken, limiting the clarity and comparability of results.
- **Data quality:** Discrepancies in the availability of municipal infrastructure, human capital, and budget were frequently recognised as limitations to the quality of waste data obtained. A lack of landfill infrastructure — notably weighbridges — was frequently cited as a limitation in the accuracy of waste quantification, necessitating both volume and density estimates.
- **Representation:** The majority of the reviewed IWMPs presented information on formal waste streams, with limited representation or consideration of informal waste streams. While it was frequently acknowledged that illegal dumping and littering constituted major challenges to the municipality, quantification of this waste was not available. Furthermore, while the percentage of un-serviced households in terms of waste management was a mandated category for reporting, information on the volume of un-recovered waste from un-serviced households was not available.

According to Sango, Basson and Williams (2016), while the development of so-called 1st and 2nd generation IWMPs are indicative of progress with regards to municipalities' approaches towards waste management, deficiencies highlighted in these reports further include compliance gaps and the lack of waste information at municipal level. It is anticipated that the focus of the 3rd generation IWMPs will be directed towards implementation plans to address the deficiencies highlighted in the earlier generation plans (Sango, Basson & Williams, 2016).

Despite these limitations, analysis of available IWMPs still provides perspective on the state of waste management in South Africa. Notwithstanding various financial and logistical challenges, in general a review of IWMPs would suggest that the country's historical dependence on landfill operations is currently undergoing a gradual shift to alternative treatment technologies, which retain the value of waste (Sango, Basson & Williams, 2016). While increasing the landfill diversion rate remains a key priority for government, the lack of adequate information on waste volume and composition (see Appendix A, Section A.4) has been recognised as a key factor in the failure of various interventions previously introduced in South African municipalities (Oelofse, Muswema & Koen, 2016). Thus, limitations in accurate waste data not only prohibits representation of the waste sector at a national level, but should further be regarded as a caveat for municipalities in the implementation of waste management strategies.

4.4 Informal and Mismanaged Waste

In South Africa, an additional shortcoming in the availability of waste data is the lack of consideration or representation of waste generated and disposed of outside of formal waste streams. The exclusion of so-called unofficial waste from national waste estimates has the potential to significantly under-estimate national waste tonnages. The focus of this section is directed towards understanding and estimating informal waste generation and management in South Africa.

4.4.1 Estimation of Informal Waste Generation

Given the lack of information regarding informal waste generation and management in South Africa, an estimate of the quantity of informally managed household waste was developed. The starting point in this analysis is mapping the provision of waste management services in South Africa using the information contained in the *General Household Survey 2015* (Stats SA, 2016a). A summary of the key results from this survey with respect to waste service provision is shown in Table 4.4.

Table 4.4 Household refuse removal in South Africa by settlement type (rural, urban and metro households) (Adapted from Stats SA, 2016a:48)

	Total	Rural	Urban	Metro
Refuse removed at least once a week	63.5%	9.6%	81%	88%
Refuse removed less than once a week	2.4%	1.0%	3.4%	2.7%
Communal refuse dump	2.9%	2.2%	2.0%	3.9%
Own refuse dump	28%	82%	10.0%	3.9%
Dump rubbish anywhere	2.8%	4.4%	3.1%	1.5%
Other	0.4%	1.1%	0.40%	0.10%
Managed waste percentage	**69%**	**13%**	**87%**	**94.5%**
Unmanaged waste percentage	**31%**	**87%**	**14%**	**5.5%**

Table 4.4 shows that there are large discrepancies between waste services provided to households in rural and urban areas and, to a lesser extent, between urban and metropolitan areas. If it is assumed that any form of regular waste collection service and centralised communal dumps constitute a formal waste management service, with the latter three categories shown in Table 4.4 constituting informal waste management, then overall, approximately 69% of households in South Africa receive some form of formal waste management service. Considering waste service delivery per settlement type (i.e. services received by households in rural, urban, and metropolitan areas) shows that 94.5% and 87% of households in metropolitan and urban areas respectively receive some form of waste management service, while in rural areas this decreases to just 13%. Following the assumption that private refuse dumps, dumping, and "other" can be considered informal waste management, the relative uptake of informal options by households in different settlement types is illustrated in Figure 4.3.

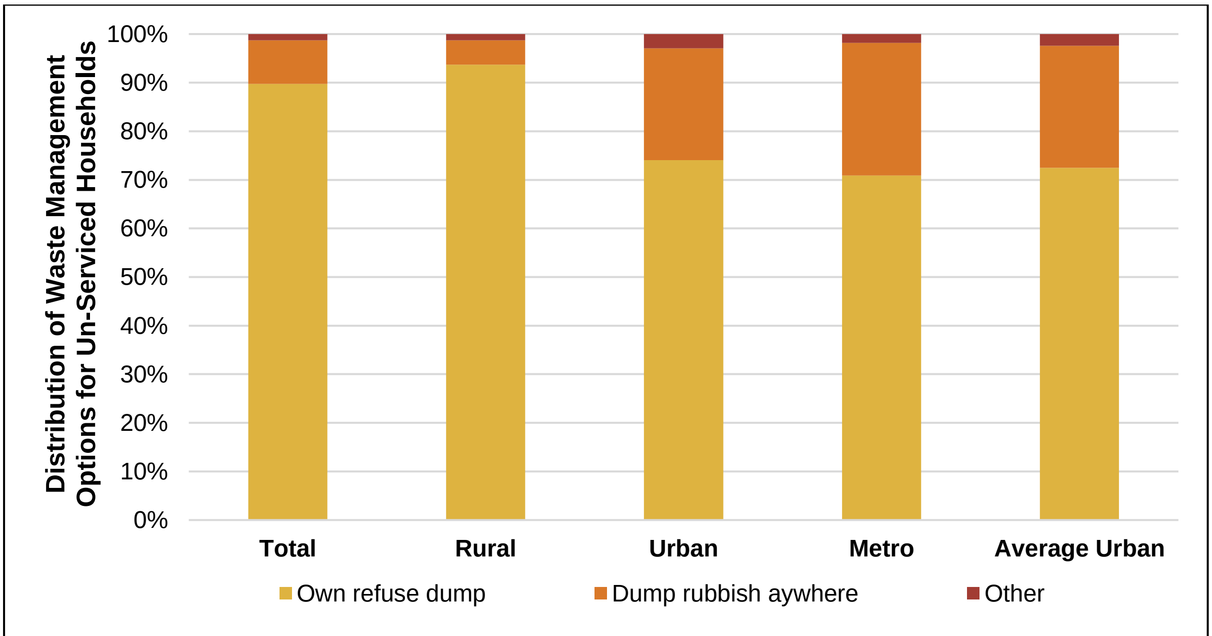

Figure 4.3 Distribution of waste management options for un-serviced South African households (Adapted from Stats SA, 2016a:48)

Figure 4.3 suggests that the most common waste management option for un-serviced households is the use of a private refuse dump. The use of private dumps by un-serviced households is most prominent in rural areas (94%) but is also relatively high in urban and metro areas (74% and 71%, respectively). Illegal dumping of household waste ranges from 5% for un-serviced rural households to 27% in metro areas, with the balance made up by "other" disposal/treatment options. Although further detail with regards to the nature of these management options is limited, it is assumed that private refuse dumps lacked any form of engineering or control (essentially an open dump) and "other" referred to treatment such an open burning.

While the results shown in Table 4.4 and Figure 4.3 provide information with regards to the service level received by different households, this can not necessarily be applied directly to waste quantities, given the regional and socio-economic differences that occur in per capita waste generation rates (see Table 4.3). This implies that although approximately 31% of South African households do not receive a formal waste management service, this does not necessarily equate to 31% of the total domestic waste generated in the country. Therefore, in order to estimate the total quantity of informal domestic waste, a model was developed to take into account differences in WGRs based on income and settlement type. Key modelling parameters regarding WGRs and national waste service delivery were extracted from Table 4.3 and Table 4.4 respectively, and used in conjunction with 2016 population statistics. WGRs were based on those provided by the DEAT (2006), and household income distribution was as reported by Stats SA (2015). The key modelling parameters considered (excluding national waste service delivery, which is available in Table 4.4) are shown in Table 4.5. It should be noted that this model is intended to provide an estimate of informal domestic waste flows and thus, a number of assumptions were used. Key assumptions used in this model are shown below.

i. Household settlement (rural, urban, and metro) distribution and income distribution based on number of households in each category could be used as a proxy for urban, rural, and metro population distribution and income distribution, even though rural households typically have a higher number of inhabitants than households in urban and metropolitan areas.
ii. 2011 and 2015 statistical data is representative of the 2016 status quo with regards to household waste service delivery and income distribution between settlement types.
iii. WGRs in no-income and low-income population groups are the same.
iv. The level of waste service delivery in rural areas is consistent across income groups.
v. In urban and metro areas, 100% of un-serviced households fall within the no-income/low-income population group.

The resulting model provides an estimate for both total domestic waste generation and total informal waste generation in South Africa. The results obtained from this model are shown in Table 4.6 overleaf. An overview of the modelling procedure is available in Appendix A, Section A.5.

Table 4.5 Key modelling parameters for waste generation and population distribution

Data				
South African Population 2016[a]	54 978 907			
	No Income	**Low Income**	**Middle Income**	**Upper Income**
National Household Distribution[b]	15.5%	29.0%	48.3%	7.3%
Rural % of income group[b]	29.9%	46.8%	27.2%	7.60%
Urban % of income group[b]	70.1%	53.2%	72.8%	92.40%
Domestic Waste Generation Rate[c] (kg/capita/day)	0.41		0.74	1.29

[a] Worldometers (2017)
[b] Stats SA (2015)
[c] DEAT (2006)

Table 4.6 Overview of the results obtained for the estimate of informally managed domestic waste in South Africa

<table>
<tr><th>Total domestic waste generation (t/a)</th><th>No Income</th><th>Low Income</th><th>Middle Income</th><th>Upper Income</th></tr>
<tr><td>Rural</td><td>381 000</td><td>1 120 000</td><td>1 950 000</td><td>1 440 00</td></tr>
<tr><td>Urban</td><td>894 000</td><td>1 270 000</td><td>5 210 000</td><td>1 750 000</td></tr>
<tr><td>Total (per income group)</td><td>1 280 000</td><td>2 390 000</td><td>7 160 000</td><td>1 890 000</td></tr>
<tr><td>Total</td><td colspan="4">12 720 000</td></tr>
<tr><th>Waste subjected to informal management (t/a)</th><th>No Income</th><th>Low Income</th><th>Middle Income</th><th>Upper Income</th></tr>
<tr><td>Rural: 87.2% of population un-serviced</td><td>333 000</td><td>974 000</td><td>1 700 000</td><td>125 000</td></tr>
<tr><td>Urban: 9.5% of population un-serviced</td><td>84 900</td><td>121 000</td><td>495 000</td><td>166 000</td></tr>
<tr><td>Total (per income group)</td><td>417 000</td><td>1 090 000</td><td>2 190 000</td><td>291 000</td></tr>
<tr><td>Total</td><td colspan="4">3 988 000</td></tr>
<tr><td colspan="5">Correction for distribution of waste services</td></tr>
<tr><td>Rural: Even proportion of un-serviced households across income groups</td><td>333 000</td><td>974 000</td><td>1 700 000</td><td>125 000</td></tr>
<tr><td>Urban: 100% of un-serviced urban households are in low/no income group</td><td colspan="2">533 000</td><td>-</td><td>-</td></tr>
<tr><td>Total</td><td colspan="4">3 665 000</td></tr>
</table>

If income-specific waste generation rates are considered, it is estimated that South Africa generates approximately 12.7 million tonnes of domestic waste per annum. This is comparable to the 8.06 million tonnes of municipal waste (GW01, non-recyclable portion) reported by the *NWBR* (DEA, 2012a) for 2011. Based on the model output and 2016 population statistics, an average per capita domestic WGR for South Africa can be estimated at 0.63 kg/capita/day. Comparison of this figure to the domestic WGRs reported by the DEAT (2006) (Table 4.5) suggests that this generation rate lies between that of low- and middle-income groups. This provides a more realistic estimate for average per capita waste generation in South Africa, as opposed to using the arithmetic mean of the domestic WGRs reported by the DEAT (2006) (0.81 kg/capita/day), which suggests that average waste generation lies between middle-income and high-income groups. Given the income inequality in South Africa, the arithmetic mean is unlikely to be representative of the actual mean, as a large proportion of the population falls below the middle-income grouping. Indeed, if the arithmetic mean is used to determine annual waste generation in South Africa, this results in an estimated 44 million tonnes of domestic waste. This is notably higher than the quantity reported by either the *NWBR* (DEA, 2012a) or estimated by means of the modelled approach. This large discrepancy illustrates the importance of accounting for socio-economic distribution and income-specific waste generation rates in a national waste estimate.

As shown in Table 4.6, accounting for both settlement type and income distribution suggests that informal household waste generation in South Africa is in the order of 3.67 million tonnes. Comparison of this figure to the 12.7 million tonnes estimated for total domestic waste generation suggests that approximately 29% of domestic waste generated in South Africa is not collected or treated via formal management options. Although the model was developed to take into account income specific WGRs and the population and income distribution of different settlement types, it was assumed that WGRs were not affected by settlement type. While Table 4.3 suggests that settlement type could be considered a factor in terms of waste generation, this observation was not applied to the model due to limitations in the availability of WGRs accounting for both income and settlement type. Therefore, the model output could be improved with further refinement of the WGRs used, however, this would require — amongst else — the development of better representative national WGRs.

In lieu of a more detailed estimate, the model result for informal domestic waste can be used to estimate the quantity of waste per management option utilised by un-serviced households, when considered in conjunction with the results shown in Figure 4.3. This result is shown in Figure 4.4. According to this figure, the use of a private dump is the most common waste management option for un-serviced households, with a total of 3.32 million tonnes of waste disposed of annually by this means.

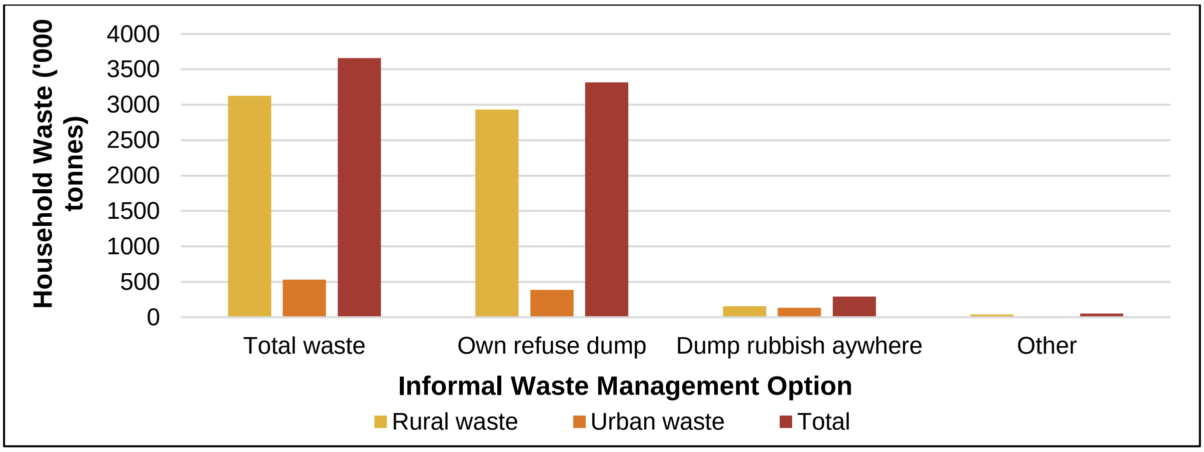

Figure 4.4 Estimated waste quantity per management option for un-serviced South African households in 2016

4.4.2 Other Sources of Informal Waste

The model developed in Section 4.4.1 provides an estimate for informal domestic waste and therefore represents only a proportion of the total informal general waste generated in South Africa. If all general waste is considered, the contribution of categories such as builder's waste (which, according to IWMPs and stakeholder engagement is the most frequently dumped material), and commercial and industrial waste is likely to increase the estimated quantity of informal waste further. Limited quantified information is available to obtain an estimate for such illegally dumped materials. Furthermore, given that the clearing of illegally dumped material is the responsibility of the municipality, such material is frequently re-incorporated into formal waste streams. Limited documentation of this recovered material further complicates the quantification of informal waste tonnages.

A similar challenge exists with regards to quantifying both recovered and unrecovered litter. Despite the extent of littering being visually obvious in many parts of South Africa, the availability of studies which focus on the quantification and characterisation of litter are limited. Available studies also tend to represent litter occurring in a relatively defined area and, as such, have limited application to the national level. While sources of litter are varied, important sources include the anti-social behaviour of individuals discarding waste directly into the environment and inadequate disposal facilities, including limitations in waste collection practices and inappropriate means for containing waste (Hall (1996) as quoted by Armitage and Rooseboom (2000:182)).

Inadequacies in certain waste disposal facilities is a further challenge in mapping waste management in South Africa. While certain facilities might receive and record municipal waste, this does not necessarily indicate that they are compliant with the required operating standards. Representing the quantity of this mismanaged waste is challenging as it is a.) unclear to what extent such facilities are represented by formal waste estimates and b.) difficult to define the standards of a site that constitute mismanagement, and hence whether such waste should be considered formal or informal. Jambeck et al. (2015) provides one of the few studies which quantifies mismanaged waste flows in South Africa. According to this study, 56% of South Africa's waste is mismanaged (of which 54% is "inadequately managed" waste and 2% "littering") (Jambeck et al., 2015:769). Based on this assumption, South Africa is ranked 11th globally in terms of its contribution towards plastic marine litter (Jambeck et al., 2015).

Using the methodology and assumptions laid out by Jambeck et al. (2015), the total quantity of mismanaged waste in South Africa in 2016 is estimated at 62 million tonnes, of which 2.2 million tonnes are litter and the balance representing inadequately disposed of waste. These estimates arguably provide an overestimate of the mismanaged waste quantity, due to the lack of clarity in the definition of "inadequately managed" waste. According to Jambeck et al. (2015:768), inadequate disposal incorporates "not formally managed waste [including] disposal in dumps or open, uncontrolled landfills". While a large proportion of South Africa's landfill sites operate with limited or negligible engineering controls, these do not necessarily warrant classification as an informal management option. While the uncertainty associated with these estimates limit their usefulness in establishing the status quo with regards to waste management in South Africa, they non-the-less highlight the importance of mapping both managed and mismanaged waste fractions when developing a national waste estimate.

4.5 Mapping General Waste Management in South Africa

Despite inconsistencies in the availability of reliable national waste quantification and characterisation data (see Appendix A.4), the findings and discussion presented in Sections 4.3 – 4.4 can be used to develop a basic mapping of general waste flows in South Africa. The resulting schematic diagram is shown in Figure 4.5 overleaf. For consistency with national waste reporting standards, Figure 4.5 uses the categorisation regulated in the *National Environmental Management: Waste Act, No. 59 of 2008. National Waste Information Regulations* (2012:s8) for waste disposal and management options[9].

It is shown in Figure 4.5 that although the initial distinction between informal and formal waste can be made relatively easily depending on whether or not waste is directly collected or transported to a waste management facility, thereafter the distinction becomes more difficult. Not only can informally dumped or littered waste be recovered by the municipality, and hence move into the formal waste stream, but formal management facilities might be unlicensed and/or non-compliant with operating standards. The potential for mismanagement challenges the accurate representation of waste flows, as unlicensed and non-compliant facilities do not necessarily report data.

Given the identified waste data uncertainties for the waste sector as a whole, Figure 4.5 is limited to providing an illustration of different waste management options and lacks an indication of the relative quantities of waste managed under each option. However, if the *NWBR* (DEA, 2012a) and SAWIS data are accepted as the most comprehensive sources of waste data currently available, then in terms of formal waste management, South Africa still reflects a strong dependence on landfill. However, the recycling rates reported by the *NWBR* (DEA, 2012a) for certain materials are suggestive of reform within this sector. Furthermore, South Africa's formal commitment to "green economy" initiatives (von Blottnitz, 2016) and recent flirtation with "circular economy" discourse (DEA, 2017a) suggest that despite the current dependence on landfill, the proportion of waste diverted from landfill will continue to increase.

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9 This categorisation is only strictly applicable to licensed waste management facilities required to register on the SAWIS in accordance with national regulations. Informal waste management facilities could fall below the minimum requirements for both licensing and reporting. and hence lack formal classification. For the sake of consistency, it was assumed that formal categorisation covers all potential waste management options and informal waste management can be classified under these categories, regardless of whether or not they meet the criteria for formal licensing or registration with SAWIS.

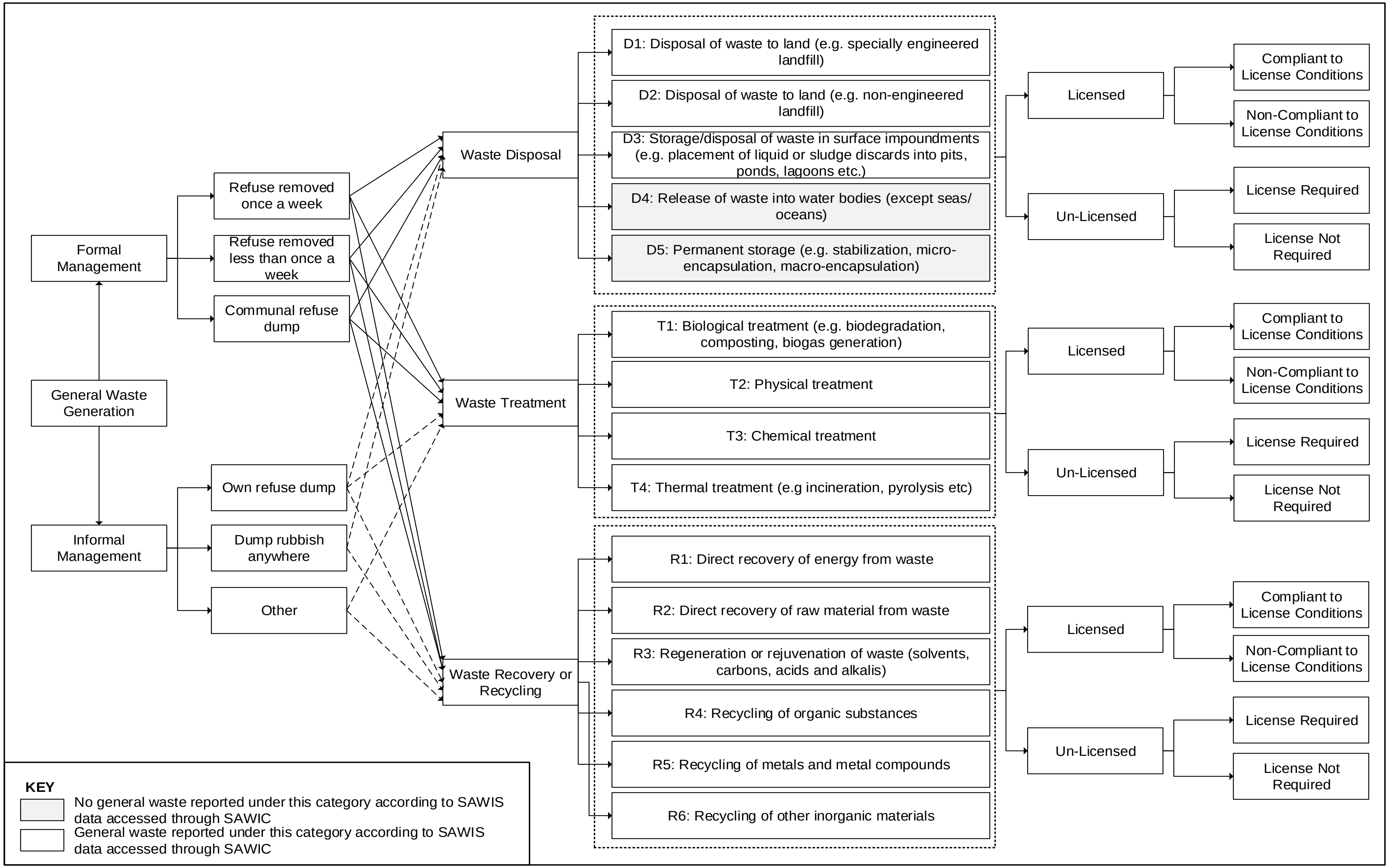

Figure 4.5 Schematic diagram providing a basic mapping of general waste management in South Africa

4.6 Landfill Disposal in South Africa

Following the results and discussion in Sections 4.3 - 4.5, landfilling is the predominant disposal method in South Africa, with 1203 recorded landfill sites in operation in 2013 (Friedrich, 2013). Despite the requirements stipulated in the *NEM:WA* (2009:chap 5), licensing of these facilities is relatively poor. According to the DEA (2016), 56.4% of general waste landfill facilities are unpermitted in terms of the existing legislation[10]. Even with government commitment to address the backlog of unlicensed landfill sites by September 2014 (DEA, 2014b), a reported 247 (20.5%) sites remained unlicensed after this deadline (News24, 2014). Although the target was not achieved, the proportion of licensed landfill sites increased to 79.5%, a near doubling in three years. Assuming similar progress, it is anticipated that further improvement in addressing this backlog has been made in subsequent years.

In terms of assessing the impacts of landfill disposal, unlicensed facilities are only part of the problem. Even for licensed sites, historically there has been limited information available with regards to sites' compliance to environmental management and pollution control standards (Godfrey, 2008). Given the variation in the level of engineering control on different sites — such as LFG collection and waste covering and compaction — the lack of stringent reporting poses a particular challenge in assessing landfill impacts at the national level (Friedrich, 2013). The lack of information regarding landfill impacts is compounded by the existence of informal and unregulated disposal sites. While on an individual basis, the impacts from such sites may be small, aggregating the impacts to the national level could become significant, with an estimated 15 000 unrecorded private and communal dumpsites in operation in South Africa (DEAT, 2006).

Given the apparent range in the operating standards of different landfill sites, representation of a South African landfill scenario for assessing the end-of-life impacts of a product in a LCA application requires an understanding of how these operations differ in terms of their impact potential. The objective of this section is to investigate the nature of landfill operations in South Africa, and provide insight into the operating standards and conditions with a particular focus on the impact potential of sites.

4.6.1 Overview of Landfill Operations in South Africa

South Africa's strong dependence on landfill for the treatment of general waste has been attributed to its historical position as the cheapest and easiest disposal method (DWAF, 1998c). Despite increasing awareness of the need for waste management in alignment with the principles of the waste hierarchy, landfill remains popular due to its low cost and relative ease of operation. While the environmental impacts of landfilling can be controlled if operations are adequately managed, many South African landfills have in the past fallen short of acceptable operating standards (DWAF, 1998c). Such shortcomings have been attributed to poor siting, design, and operation, which presents a significant potential for pollution (DWAF, 1998c). In order to promote compliance with environmental policy and legislation, a set of documents outlining the minimum requirements for waste disposal were developed (Bhailall, 2015). The three key documents that outline the minimum requirements are as follows:

1. *Minimum Requirements for Waste Disposal by Landfill* (DWAF, 1998c)
2. *Minimum Requirements for the Handling, Classification and Disposal of Hazardous Waste* (DWAF, 1998a)
3. *Minimum Requirements for Water Monitoring at Waste Facilities* (DWAF, 1998b)

[10] This figure is based on statistics reported by the DEAT (2005) (as quoted in DEA (2011a) and hence cannot necessarily be considered reflective of the status quo with regards to licensing in South Africa. However, no more current statistics were available.

Due to the variation in the size and nature of landfill sites across the country, the *Minimum Requirements for Waste Disposal by Landfill* (DWAF, 1998c) recognise different classes of landfill for which different regulations are prescribed. An initial distinction imposed on a landfill is based on the type of waste it accepts (general or hazardous). Thereafter, landfill classification is made with regards to the size of the site and its potential to generate leachate. The former classification is based on the maximum rate of deposition (MRD) (tonnes/day) at the site, and the latter is determined from a simplified climatic water balance, where B^+ signifies the potential for significant leachate generation and B^- signifies sporadic leachate generation potential. An overview of the classification of general waste landfills, as stipulated in the *Minimum Requirements for Waste Disposal by Landfill* (DWAF, 1998c:s3), is shown in Table 4.7.

Table 4.7 Landfill site classification for general waste as laid out in the Minimum Requirements for Waste Disposal by Landfill (DWAF, 1998c:s3).

Landfill Size Class	**C**		**S**		**M**		**L**	
	Communal		Small		Medium		Large	
Legend	B^+		Potential for significant leachate generation					
	B^-		Sporadic leachate generation likely					
	G		Landfill accepting general waste					
	MRD		Maximum rate of deposition (tonnes/day)					
MRD	<25		>25, <150		>150, <500		>500	
Climatic Water Balance	B^-	B^+	B^-	B^+	B^-	B^+	B^-	B^+
Code	G:C:B^-	G:C:B^+	G:S:B^-	G:S:B^+	G:M:B^-	G:M:B^+	G:L:B^-	G:L:B^+

Subsequent to the publication of the Minimum Requirements, driven by increasing environmental awareness and the promulgation of the *NEM:WA* (2009), additional legislation such as the *National Environmental Management: Waste Act, No 59 of 2008. National Norms and Standards for the Disposal of Waste to Landfill* (2013) has been passed, outlining more stringent requirements for landfill operations. Whilst the landfill classification system shown in Table 4.7 is still applicable to landfills in South Africa, Section 3 and Section 4 of these National Norms and Standards prescribe new landfill classes (Class A, B, C, D) with more advanced liner designs, and stricter waste acceptance criteria and disposal restrictions (Bhailall, 2015). Despite these new classification and construction requirements, the *Minimum Requirements for Waste Disposal by Landfill* (DWAF, 1998c) are still applicable for the operation and monitoring of landfill sites (Bhailall, 2015).

The objective of landfill regulations is to manage and reduce the environmental impact associated with landfill operations. Therefore, classification of landfill sites based on their size and leachate generation potential affords the opportunity to identify the most relevant regulations and controls for each site, as opposed to stipulating one set of regulations that might not be of relevance to all sites. For example, for sites generating sporadic leachate, the establishment of a costly leachate management system is of less relevance than it would be to a site where significant leachate generation requires strict control (DWAF, 1998c). It is therefore questionable as to whether, for the purposes of evaluating the end-of-life impacts from a LCA perspective, different landfill sites can be represented by a generic landfill dataset, or whether further distinctions should be made with regards to the nature and distribution of site types. For example, given the variation in the landfill sites operating in South Africa, the best representation of disposal might be by means of a "market type" dataset with different types of sites taking different proportions of the disposed waste.

4.6.2 Distribution of General Waste Between Different Landfill Classes

The quantification of the waste flows received by each landfill class provides an indication of the relative importance of each type of landfill site within the context of South Africa's formal waste management system. Due to limitations in the availability of disposal tonnages for all landfill sites in South Africa disaggregated to the level of landfill size class as shown in Table 4.7, an estimate was developed for the amount of general waste accepted by landfill sites falling under each landfill class. This estimate was developed from a list of licensed landfill operations in the country (available from the SAWIC (DEA, 2017b)), which was screened to isolate sites accepting general waste. The resulting list was then sorted according to landfill size classification. This list provides a record of licensed facilities but does not contain any information regarding daily waste flows received by each site. The relative quantity of general waste treated in each size class was therefore estimated based on the following assumptions:

i. Cumulative distribution function of sites between landfill size classes can be represented by a lognormal distribution.
ii. The conditional expected value for the waste deposition for each size class (tonnes/day) was determined from the resulting lognormal probability distribution function.
iii. The maximum waste flow for the large sites was taken as 5 000 tons/day based on the average deposition reported to Bisasar Road, which was reportedly the largest landfill site operating in South Africa. This is however likely to represent an overestimate.
iv. The approximate daily waste flow for each size class was calculated as the conditional expected value (daily flow) per site size class multiplied by the number of sites within the particular size class.

Variation in the estimated waste deposition rate for any landfill size class has a direct impact on both the relative contribution from each size class and the total waste tonnages. It should therefore be noted that this approach is intended to provide an estimate of the relative quantity of waste received by different landfill size classes, as opposed to a detailed study, and thus the results obtained must be regarded as a broad estimate providing a basic mapping of waste flows. A more detailed study would be required for improved accuracy. The results obtained from this approach are shown in Table 4.8. Further detail on the approach taken, the estimated daily waste flows assuming a lognormal distribution, and determination of the possible range is available in Appendix A, Section A.6.

Table 4.8 Estimate of the distribution of general waste between landfill classes in South Africa

Size Class	Waste Distribution (% of Total Flow)	Possible Range[a] in Waste Distribution (% of Total Flow)	Sporadic Leachate Generation (% B^- Sites per Size Class)	Licensed Sites per Size Class (% of Total Sites)
C: Communal	2.8	2.0 – 3.0	90	50
S: Small	12	9.4 – 11	84	27
M: Medium	31	19 – 40	70	16
L: Large	54	47 – 68	47	5.8

[a] See Appendix A, Table A.11

Based on this method, the total general waste deposited into licensed landfills is estimated to be in the order of 44.5 million tonnes/year (see Table A.10, Appendix A). This result lies between that reported to SAWIS (36.2 million tonnes) and that extrapolated from the *NWBR* (DEA, 2012a) (53.5 million tonnes). The results shown in Table 4.8 suggest that the majority of licensed landfill sites operating in South Africa fall under the C-Class size classification, with the proportion of licensed sites within a size class decreasing with increasing size class. However, due to the small volume of waste accepted at each communal site (C-Class), the estimated proportion of waste managed at each type of site reflects the inverse trend, with sites falling within the L-Class accepting the majority of waste. According to the range defined in Table 4.8, the relative contribution of large landfill sites towards landfill management ranges between 47 – 68%. It could therefore be assumed that despite the relatively small number of licensed L-Class sites, this landfill type best represents the management of general waste in South Africa.

Given that these results are based on licensed landfill sites only, it is anticipated that the inclusion of private disposal sites (unlicensed in terms of Section 9 of the *National Environmental Management: Waste Act, No. 59 of 2008. Waste Classification and Management Regulations* (2013)) would increase the relative number of communal sites operating in South Africa and hence the proportion of waste disposed of within this size class. According to the *General Household Survey 2015* (Stats SA, 2016a), approximately five million households in South Africa are un-serviced, of which the majority dispose of waste in a private dump. Based on the model developed in Section 4.4 to quantify this informal domestic waste, approximately 3.32 million tonnes of general waste are disposed of in private refuse dumps. If this quantity is included in the estimated yearly waste deposition for communal landfill sites (Table 4.8) and a new distribution determined, the relative contribution of this size class increases to 10%.

The quantity of waste managed at each type of site does not necessarily reflect the environmental impact associated with the operation of the site. The impact potential of the site is dependent on a number of parameters including the waste type landfilled and site-specific parameters, such as the barriers and controls in place to manage the emissions from the waste body. LFG and leachate generation are the two primary sources of emissions. While both of these are recognised as potential environmental hazards, leachate generation in particular is considered an important parameter influencing the environmental risk associated with a particular site (DWAF, 1998c). As shown in Table 4.7, the leachate generation potential of the site (denoted by B^+ and B^-) is included within the landfill site classification system. Where a site is classified as B^+, more stringent management requirements are imposed on the site in order to control the risk associated with potentially significant volumes of leachate (DWAF, 1998c). According to Table 4.8, for smaller landfill sites, the majority fall under the B^- classification, unlike the large size class where there is an approximately even split.

4.6.3 National Standards and Requirements for Different Landfill Classes

As noted in Section 4.6.1, the operation of landfill sites is informed by a set of minimum requirements. The objective of these minimum requirements is to manage and control any adverse environmental risks associated with landfill operations. The minimum requirements thus define the necessary infrastructure, operating conditions and technology for a site for the management and containment of emissions. In order to assess the potential environmental impacts arising from a site, it is necessary to understand the level of environmental controls implemented on each site.

While the *National Environmental Management: Waste Act, No 59 of 2008. National Norms and Standards for the Disposal of Waste to Landfill* (2013) and the *Minimum Requirements for Waste Disposal by Landfill* (DWAF, 1998c) provide a comprehensive source of construction, operating and monitoring standards with regards to landfill operations, not all of these standards are of relevance in understanding the environmental impact potential of different sites. To determine the applicability of existing end-of-life models and datasets to representing waste disposal in South Africa, regulations and standards relevant to modelling the emission output and impact potential from a landfill site were isolated. Standards and regulations with no direct impact on the physical emission output from the site were not considered. An overview of the design and operating regulations identified for their potential influence on landfill emissions are shown in Table 4.9 overleaf and Table 4.10 (pg.76) respectively.

Table 4.9 Comparison of the minimum design requirements pertaining to different landfill classes Adapted from DWAF, 1998c:4-14, 8-13 – 8-17)

Landfill Class	**C**		**S**		**M**		**L**	
	Communal		Small		Medium		Large	
	B^-	B^+	B^-	B^+	B^-	B^+	B^-	B^+
Legend								
R Requirement								
N Not a requirement								
F Flag: special consideration to be given by expert or departmental representative								
B^+ Potential for significant leachate generation								
B^- Sporadic leachate generation likely								
Minimum Requirements for Site Selection								
Buffer Zone (m)	200	200	400	400	F	F	F	F
Minimum unsaturated zone (m)	2	2	2	F	F	F	F	F
Minimum Requirements for Landfill Design								
Conceptual Design								
Design of leachate management system	N	N	N	R	N	R	N	R
Design of toe drains	N	R	N	R	R	R	R	R
Monitoring system design	N	N	F	R	R	R	R	R
End-use plan	N	N	R	R	R	R	R	R
Testing of soils and materials	N	N	N	F	F	F	F	F
Technical Design								
Surface hydrology and drainage design	N	N	N	F	R	R	R	R
Water quality monitoring system	N	F	N	R	R	R	R	R
Leachate detection system	N	F	F	N	R	N	R	N
Leachate treatment system	N	N	N	F	N	R	N	R
Leachate management and monitoring system	N	F	N	R	N	R	N	R
Gas management and monitoring system	N	N	N	N	F	F	F	F
Stability of slopes	N	N	F	F	F	F	F	R
Erosion control design	N	N	F	F	R	R	R	R
Minimum Requirements for Liner Components								
Desiccation protection	N	N	N	N	R	N	R	N
Leachate collection layer	N	N	N	R	N	R	N	R
Secondary compacted clay liner	N	N	N	N	N	R	N	R
Geotextile layer	N	N	N	N	N	R	N	R
Leakage detection layer	N	N	N	N	N	R	N	R
Primary compacted clay liner	N	N	N	R	R	R	R	R
Base preparation layer	N	N	R	R	R	R	R	R
Minimum Requirements for Capping Components								
Compacted clay layer	N	N	R	R	R	R	R	R
Geotextile layer	N	N	N	N	N	R	N	R
Gas Drainage Layer	N	N	N	N	N	R	N	R

Table 4.10 Comparison of the minimum operating, closure and water monitoring requirements pertaining to different landfill classes (Adapted from DWAF, 1998c:10-17 – 10.18, 12-7, 13-5)

Landfill Class	**C**		**S**		**M**		**L**	
	Communal		Small		Medium		Large	
	B^-	B^+	B^-	B^+	B^-	B^+	B^-	B^+
Legend								
R Requirement								
N Not a requirement								
F Flag: special consideration to be given by expert or departmental representative								
B^+ Potential for significant leachate generation								
B^- Sporadic leachate generation likely								
Minimum Requirements for Landfill Operation								
Landfill Operation								
Compaction of waste	N	N	R	R	R	R	R	R
Daily cover	F	F	R	R	R	R	R	R
End-tipping prohibited	N	N	N	N	R	R	R	R
Waste burning prohibited	F	F	F	F	R	R	R	R
Contaminated run-off contained	F	F	F	F	R	R	R	R
Leachate contained	N	F	F	R	F	R	F	R
0.5m freeboard for diversion and impoundments	F	F	R	R	R	R	R	R
Landfill gas control	N	N	F	F	F	F	F	F
Minimum Requirements for Rehabilitation, Closure and End-use								
Design anti-erosion measures	F	F	R	R	R	R	R	R
Ongoing leachate management	N	N	F	R	F	R	F	R
Ongoing gas management	N	N	F	F	F	F	F	F
Ongoing inspection and maintenance	N	N	R	R	R	R	R	R
Minimum Requirements for Water Quality Monitoring								
Operation Monitoring								
Surface water monitoring	F	F	F	R	R	R	R	R
Groundwater monitoring	N	F	R	F	R	R	R	R
Leachate monitoring	N	F	N	R	N	R	N	R
Post Closure Monitoring								
Post-closure surface water monitoring	N	F	N	R	F	R	R	R
Post-closure ground water monitoring	N	F	N	R	F	R	R	R

Table 4.9 and Table 4.10 shows that a number of key differences exist in the minimum requirements imposed on different landfill classes in South Africa. For larger landfill size classes, compliance with minimum operating standards is a regulatory requirement for the majority of standards. Smaller landfills by contrast, are largely non-regulated in terms of the conditions shown in these tables. For leachate management, Table 4.9 shows the different requirements for leachate collection and treatment systems, landfill liner and run-off containment for different sites. LFG management reflects a similar lack of consistency with regards to regulatory requirements. These differences have the potential to affect both the emissions from the site and environmental impacts thereof.

In terms of representing South African waste disposal practices within the context of LCA, given the variation that exists in the operating standards of landfill sites, the application of a generic landfill model or dataset to represent landfill emissions might be challenging. Representation of the South African landfill scenario would require that the model or dataset be capable of accurately representing various degrees of containment and treatment efficiencies, in addition to different emission outlets (i.e. leachate seeping into groundwater in the absence of a liner). The use of a generic model or dataset is further challenged by differences in the operating requirements of each landfill site. For example, operations such as compaction and covering influence the rate and extent to which waste decomposes, which in turn influences the emission generation and release from the site. Practices such as open burning introduce an additional set of emissions from the landfill operation, which would have to be considered in the South African scenario in order to accurately reflect all potential impacts.

4.7 Landfill Compliance

In terms of modelling landfill impacts within the context of LCA, while the use of minimum requirements and landfill legislation are useful in constructing models of how different landfill sites are expected to operate, the extent to which this can be considered representative of actual operations is dependent on whether landfill sites are compliant with these regulations and standards. It is a requirement stipulated in the *NEM:WA* (2009:chap 5) that every landfill site be licensed and that compliance to these license conditions be audited on a regular basis. While the number of unlicensed landfill sites in South Africa has declined sharply, this poses the question as to whether license conditions are being adhered to.

The focus of this section is directed towards assessing landfill compliance in South Africa. The objective of this analysis is to understand firstly the extent of non-compliance of landfill sites to their license conditions and secondly where non-compliance is most likely to occur.

4.7.1 Assessing Landfill Compliance

One of the license conditions for landfill disposal facilities is that regular internal and external audits of the site operations be undertaken. The results of these audits can be used to determine how closely landfill facilities comply with their license requirements. This is beneficial in developing a representative model for landfilling in South Africa and provides an indication of which, if any, non-compliances should be incorporated into this model. Monitoring the performance of unlicensed landfill sites presents a challenge, as without a license, there are no operating standards or regulations against which the performance of the site can be assessed. Furthermore, without a license, audits are not compulsory, and thus collecting and interpreting data from such a site is challenging.

The analysis in this section is limited to licensed landfill facilities. This analysis is undertaken using the landfill audit data of licensed waste disposal facilities (WDFs) in the Western Cape, provided by the Western Cape Government Environment Affairs and Development Planning Department of Waste Management (hereafter, the Department). This data includes the external audit reports conducted on licensed WDFs within the Western Cape over the course of the 2015/2016 year. A limitation in this approach is that this data is representative of landfill disposal within one province, as opposed to the country as a whole. Given that the Western Cape represents the second highest contributor to South Africa’s municipal waste stream (DEA, 2012a), analysis of the compliance within the Western Cape provides a useful starting point with regards to assessing where non-compliance in landfill operations is likely to be observed. However, it has been suggested that waste management in the Western Cape is of a relatively high standard by comparison to the rest of the country, in terms of both number of licensed sites and the compliance to permit conditions (Hanekom, personal communication 2016, 30 May 2016). The implication of this being that although a relatively high proportion of provincial disposal sites are represented by the Western Cape audit data, analysis thereof might overestimate the extent of compliance within the country as a whole.

4.7.2 Compliance of Waste Disposal Facilities in the Western Cape

The compliance of landfill sites to their license conditions was assessed with the objective of determining where – or indeed whether – non-compliance occurs with an appreciable frequency. WDFs were assessed according to the compliance rating determined for each site by the Department. The compliance status and classification system used by the Department is shown in Table 4.11.

Table 4.11 Compliance status determination and classification (As according to the Western Cape Government Department of Environmental Affairs and Development Planning)

Compliance Status	Overall Compliance Rating	Status Indicator	Action
Compliant	85 – 100%	Green	Minor improvements required
Partially compliant	65 – 84%	Amber	Improvements required
Noncompliant	0 – 64%	Red	Major improvements required

Analysis of the Western Cape landfill audit data was somewhat challenged by the nature of the license conditions and auditing process. For example, different sites have different waste licenses, thus slightly different conditions and requirements to comply with. Consequently, not all conditions are consistently audited, and the compliance score is based on the total of the audited conditions. A high compliance rating therefore does not necessarily mean that the site is compliant to all license conditions — some of the non-audited conditions could have reflected non-compliances had they been assessed. Finally, in assessing the overall compliance score, equal weighting is assigned to each condition, thus no distinction is made with regards to the severity of the condition nor its potential impacts.

The overall compliance of 66 WDFs in the Western Cape were assessed based on the overall compliance score determined for each site by the Department. These were grouped according to their subsequent compliance rating (Table 4.11). The results from this analysis are shown in Figure 4.6.

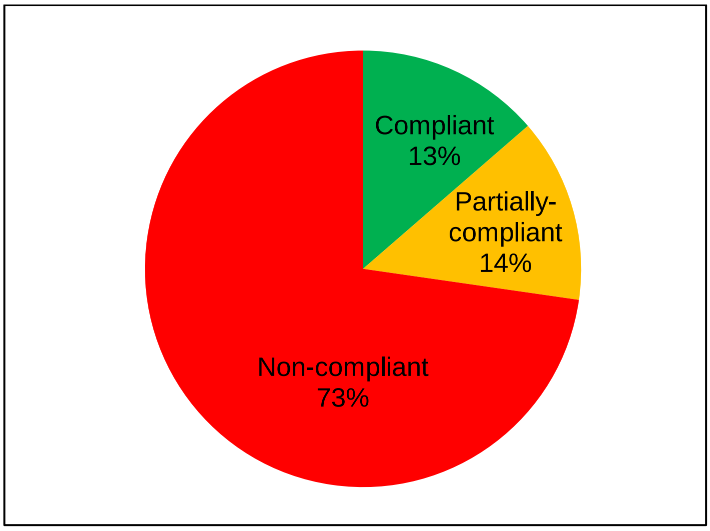

Figure 4.6 Compliance statuses (As reported to the Western Cape Government Department of Environmental Affairs and Development Planning) of 66 waste disposal facilities in the Western Cape

According to Figure 4.6, the majority (73%) of licensed WDFs within the Western Cape are non-compliant with their license conditions. This result reflects the general performance across all landfill classes. The implications of this result in terms of potential environmental impact depends on the performance of individual size classes and the specific license conditions for the non-compliant sites. A breakdown of the compliance rating within each landfill size class is shown in Table 4.12.

Table 4.12 Compliance statuses of the licensed WDFs in the Western Cape according to landfill size classification

Compliance Status	**Landfill Class**				
	G:L:B	**G:M:B**	**G:S:B**	**G:C:B**	**Unspecified**
Legend	G	Landfill accepting general waste			
	C	Communal size class			
	S	Small size class			
	M	Medium size class			
	L	Large size class			
Compliant	3	2	3	0	1
Partially-compliant	0	1	4	3	1
Non-compliant	0	3	20	16	9
Total No. of audits	3	6	27	19	11

The results in Table 4.12 suggest that within the Western Cape, larger landfill sites have a higher rate of compliance than smaller sites. All of the audited G:L:B sites are compliant with their license conditions, while the proportion of non-compliant sites increases with decreasing size class. This result suggests that a landfill model representing a G:L:B landfill site could be developed from landfill regulations, as the actual operation appears to comply with the expected standards. For smaller sites however, the frequency of non-compliance suggests that a model would need to be capable of representing the reality of the operation as opposed to the expected performance.

It should be emphasised that not all of the regulations against which compliance is audited will necessarily have a direct impact on the performance of a particular landfill site in terms of its environmental impact. The various non-compliances reported for the audited sites were analysed in order to identify those with the potential to impact on the emissions and environmental impact of the sites, and the frequency with which each non-compliance occurred. The results of the initial screening identifying the most frequent occurrences of non-compliance with the potential to influence emissions and consequent environmental impacts are shown in Table 4.13 overleaf.

According to Table 4.13, in terms of construction, inadequate leachate management, stormwater and run-off management facilities are the most frequently recorded incidences of non-compliance occurring across the audited landfill sites. Under well-managed, sanitary conditions, leachate is collected and treated prior to discharge, reducing the contaminant load and consequent impact potential thereof. When poorly managed or contained, leachate containing a full pollutant load has the potential to contaminate soil and various fresh water sources, such as groundwater or, depending on proximity and run-off volume, rivers and lakes. Other non-compliances in terms of construction include inadequate buffer zones and poor construction standards of the site. While maintaining an adequate buffer zone presents a challenge due to encroaching urban sprawl, increasing proximity of human development to a landfill site increases the risk associated with a number of emissions particularly in terms of human health. Poor construction standards increase the potential for the release of landfill emissions, thus increasing the associated risk.

In terms of operating standards, non-compliance is most frequently associated with inadequate control measures on site. This lack of control includes inadequate waste management practices in terms of compaction and covering of the waste body, unregulated management of waste (open burning), and limited control over both dust and litter from the site. Access control to the site was another area in which compliance was poor, resulting in the presence of waste pickers on site. While the presence of waste pickers does not have a significant impact the performance of the site per se, it increases the proximity of humans to the source of emissions, thus increasing human exposure to hazardous aspects of the site.

Table 4.13 Licence conditions with a high frequency of non-compliance based on Western Cape landfill audit data

	Condition of Non-Compliance
Construction	Insufficient size, structure and/or management of evaporation dam and stormwater trenches to manage precipitation during the maximum rainfall event
	Insufficient/lacking leachate management facilities
	Insufficient/lacking stormwater management facilities
	Insufficient/lacking run-off water management facilities
	Insufficient/lacking drainage facilities
	Unstable slopes
	Inadequate buffer zone
	Construction not in accordance with recognised civil engineering practice
Operations	No moveable fences. Litter was seen beyond the boundaries of the facility.
	Waste pickers evident seen on site/evidence of waste reclamation
	Burning of waste was taking place on site
	No dust control measures in place
	Inadequate compaction and covering of waste (including inadequate cover material, infrequent covering and compaction and evidence of uncovered waste)
	Landfill gas not monitored
Monitoring and Recording	There were no boreholes located at the facility
	No groundwater and surface water monitoring taking place at the facility
	No detection monitoring taking place at the facility
	No water quality tests taking place at the facility

Whilst monitoring and recording requirements cannot strictly be considered to impact on the emissions or environmental impact of a site, the frequency with which non-compliance to monitoring related license conditions occurs promotes their inclusion in the analysis. The lack of monitoring results imposes a serious limitation on the availability of data to develop a representative model of South African landfilling operations, particularly when considered in conjunction with the various non-compliance issues associated with the construction of the site. For example, without data detailing the consequences of inadequate leachate management in terms of groundwater concentrations of certain pollutants, this cannot be accurately incorporated into a model.

The frequency with which an incident of non-compliance occurs is not necessarily indicative of the permanent state of operation of the site. The objective of the audit report is to identify where a site deviates from its license requirements, and prescribe a corrective action. Thus, under ideal circumstances, the non-compliance issues identified in Table 4.13 would be corrected, leaving each site at full compliance. This reality is not necessarily observed, thus suggesting that while a current model, especially for smaller sites, might need to reflect the consequences of non-compliance, within the longer term, this need could be alleviated as all sites are brought to full compliance.

4.8 Implications of Findings for Mapping Product End-of-Life

The results and discussion presented in this chapter shows that in South Africa, the detailed mapping of quantified waste flows is prohibited by the discrepancy in waste management practices that occur, and the lack of reliable and representative waste data at the national level. Although general waste disposal data is lacking, for most recyclable materials it is possible to represent the recovery thereof. Recycling rates for commonly recycled materials (i.e. plastic, paper, cans etc.) are typically determined independently and reported by industry bodies such as Plastics SA, Packaging SA, and the Paper Recycling Association of South Africa (PRASA) or Producer Responsibility Organisations (PROs), such as Collect-a-Can, PETCO, POLYCO, SAVA and the PSPC. While accurate information on recycling rates are not necessarily available in national waste repositories, this information is likely to be available from alternative sources. Consequently, the recycled proportion of most common materials can be defined in a waste scenario with relative confidence. However, as no standard basis exists against which to define recycling rates, this introduces some complications into quantifying the proportion of a waste that is actually recycled. Assuming that alternative treatment practices, such as recycling, are typically standardised procedures for which South Africa has good infrastructure, it is anticipated that these practices are likely to be well represented within existing LCA capacity.

It is representing the fate of the remaining proportion of general waste in South Africa that presents a particular challenge to end-of-life modelling. The large inequality in the level of waste service provision suggests the importance of representing both formal and informal management practices. While it is true that for both formal and informal management the majority of waste is landfilled, there is a large discrepancy in the conditions between formal and informal disposal sites, with conditions ranging from sanitary landfill sites to open dumps. If different site conditions are considered in a representative LCA end-of-life scenario, it is necessary to define the proportion of waste disposed of under different landfill conditions. For domestic waste, assuming that informal disposal in private refuse dumps accounts for 26% of the total domestic waste generated in the country (see Section 4.4.1) and that the general waste distribution between landfill size classes (see Section 4.6.2) is applicable to domestic waste, the fate of the non-recycled portion of domestic waste in South Africa is illustrated in Figure 4.7.

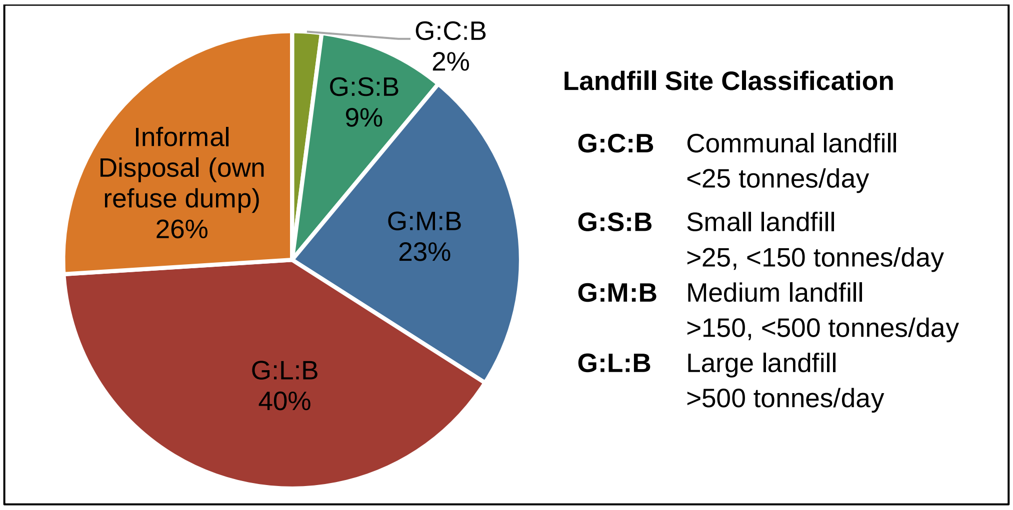

Figure 4.7 Fate of the non-recycled portion of domestic waste in South Africa (where the percentage represents the proportion of total household waste disposed of in each general waste landfill size class)

The categorisation shown in Figure 4.7 considers only the relative size class for the formal disposal sites, excluding consideration of the leachate generation potential of sites. This is an important characteristic of landfill sites, as it not only incurs differences in the regulatory requirements for the site but represents an important source of emissions. Therefore, it is of interest to disaggregate the categorisation shown in Figure 4.7 further to represent the leachate generation potential of sites. This is illustrated in Figure 4.8 overleaf.

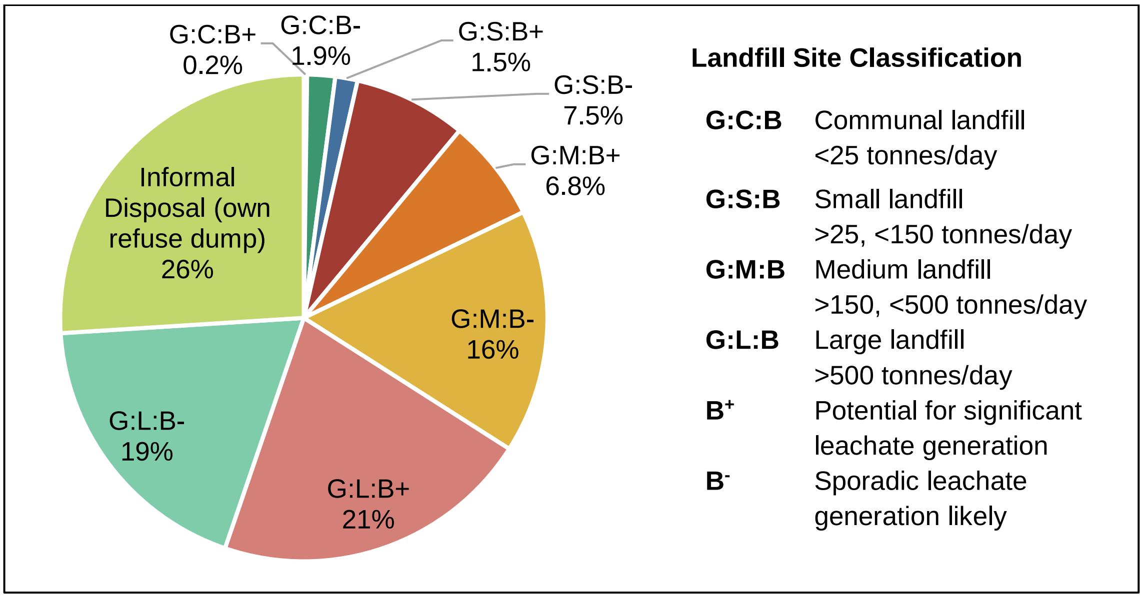

Figure 4.8 Fate of the non-recycled portion of domestic waste in South Africa taking into account different site categorisation including leachate generation potential (where the percentage represents the proportion of total household waste disposed of in each landfill class)

The benefit of Figure 4.8 is that it provides a relatively high level of disaggregation with regards to the expected conditions for different landfill disposal sites. However, given the frequency with which non-compliance to regulatory requirements is observed on South African landfill sites, further disaggregation may be necessary to represent the actual conditions of different sites. Disaggregating the landfill site categorisation further so as to account for the compliance status of sites is challenging, requiring landfill audit data at a national scale. It is the objective of the audit process to identify non-compliance issues to be addressed by the site within a stipulated time frame. Assuming that sites address any compliance issues highlighted in the audit, the occurrences of non-compliance is anticipated to decrease. It should further be considered that addressing the highlighted issues does not preclude the potential for other non-compliances to arise. This implies that mapping the occurrence of non-compliance demands a regular analysis of compliance reports.

Thus, a representation of the current compliance distribution of sites can only be regarded as a snap-shot in an evolving system. Figure 4.9 (overleaf) represents such a snap-shot of the compliance status of landfill sites based on 2015/2016 audit data. An explanation of the landfill site classification used in this figure is available in Table 4.7, Section 4.6.1. It should be noted that this representation is based on the audit report data obtained from the Western Cape DEADP (see Section 4.7), and thus assumes that the Western Cape compliance can be considered representative of the national distribution. Given the relatively high level of waste management services within the Western Cape compared to the country as a whole, the accuracy of this approach is questionable.

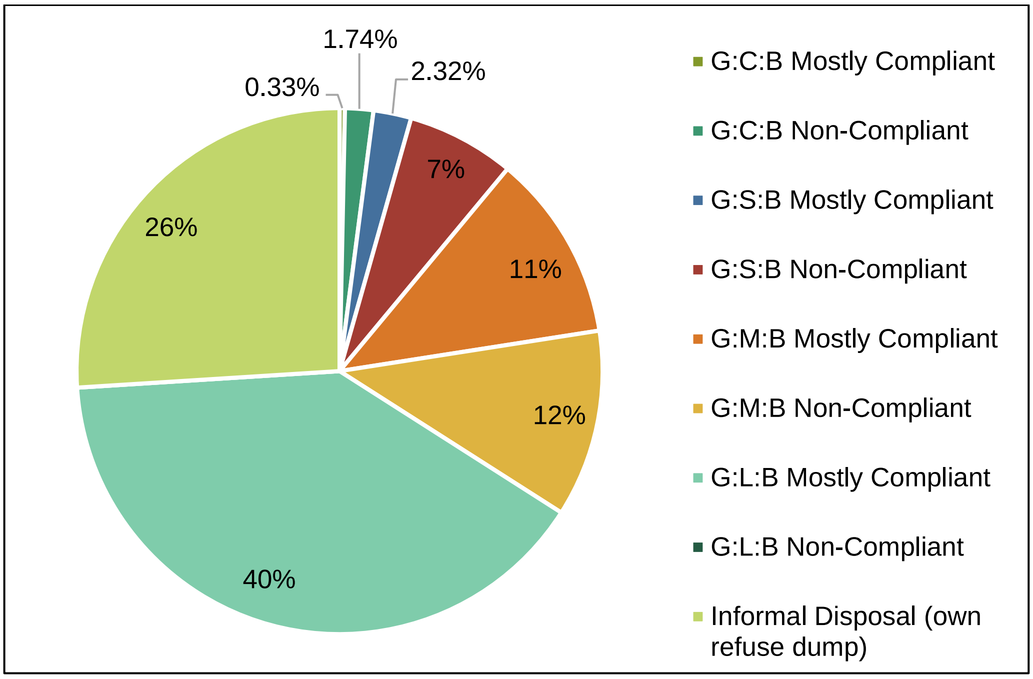

Figure 4.9 Fate of the non-recycled portion of domestic waste in South Africa taking into account different site categorisation and the compliance statuses thereof (where the percentage represents the proportion of total household waste disposed of in each landfill class).

Analysis of Figure 4.9 suggests that even if the compliance statuses of landfill sites in the Western Cape are used as a proxy for national compliance, the proportion of waste disposed of under non-complaint landfill conditions is appreciable. According to the results presented in Table 4.13, various deviations from regulated operating, construction and monitoring conditions are observed with the potential to influence the emissions and hence the environmental impact of landfill sites. While accurately quantifying the effect of non-compliance is a challenge, including these effects is of potential relevance to end-of-life modelling.

4.9 Summary

This chapter intended to establish the status quo for general waste management in South Africa and was guided by key question i (What are the current waste management practices and “market share” of each for municipal waste in South Africa, and do they vary significantly from those in developed countries?).

Key findings arising from this chapter showed that the accurate quantification and mapping of waste flows in South Africa is prohibited by the lack of comprehensive and accurate waste data. Given the disparity in the level of waste services received by South African households (only 69% of households receive a formal waste service), accurately quantifying waste flows requires the consideration of waste managed in both the formal and informal sector. However, not only does available waste data exclusively represent waste quantities managed within the formal sector, but does so with a questionable accuracy. The large proportion of un-serviced households therefore implies that existing waste quantities are likely to severely underestimate the total waste quantities generated and managed in South Africa.

Given this limitation, the purpose of the model development undertaken in this chapter was to provide an improved estimate of waste quantities managed in South Africa considering both the formal and informal sector. The resulting model utilised South African population distribution data and regional

WGRs to estimate waste quantities in a manner sensitive to the influence of both settlement type and income grouping on waste generation. The results of this model estimated that South Africa generates approximately 12.7 million tonnes of domestic waste per annum. Of this, 3.67 million tonnes are generated by informal households. This implies that 29% of domestic waste generated in South Africa is not collected or treated via formal management options. Of this waste, approximately 3.13 million tonnes (85%) is generated in rural areas. For all settlement types, the most common waste management option for un-serviced households is a private dump (the use of private dumps for the disposal of informal waste is reportedly 94%, 74%, and 71% in rural, urban, and metro areas, respectively). Illegal dumping is the next most common option and ranges from 5% for un-serviced rural households to 27% in metro areas with the balance made up by “other” disposal/treatment options.

Including the contribution of the informal sector enables the development of a better representation of the fate of general waste in South Africa, including an improved quantification of the “market share” of waste managed via different management options. For both formal and informal general waste, disposal to land (landfill and dumping) represents the most utilised waste management option. However, in terms of quantifying or assessing the impact potential of this management option, the notable variation in conditions across different disposal sites in South Africa means that it is necessary to consider the environmental impact potential of different sites, as opposed to basing an assessment on one generic set of landfill conditions. While it can be assumed that private dumps utilised in the informal sector lack engineering controls, even within the formal sector, there is variation in both the regulated site conditions as well as compliance to these conditions.

Based on the results of this chapter, approximately 40% of general domestic waste is disposed of into G:L:B sites, which typically reflect a high compliance to regulated conditions. The remaining proportion of waste is distributed between smaller formal and informal sites, both of which typically lack the level of management and control characteristic of large, well-managed sites. Therefore, disaggregating the category “disposal to land“ and quantifying the relative proportion of waste being received by different disposal sites is a necessary first step towards understanding and quantifying the environmental impact potential of waste disposed of in South Africa.

Chapter 5

REPRESENTING PRODUCT END-OF-LIFE IN SOUTH AFRICA WITH LCA

For both formal and informal waste management, the status quo findings confirm that South Africa retains a strong dependence on landfill, with the majority of the country's general waste disposed of into landfill sites with varying levels of engineering controls. In terms of modelling the end-of-life impacts of a product consumed and disposed of in South Africa, it is therefore suggested that a range of formal and informal landfill sites be considered.

The purpose of this chapter is to assess the extent to which existing LCA datasets available within SimaPro v8.3 are representative of landfill disposal in South Africa. First, an overview of the datasets available in SimaPro for modelling landfill disposal is provided. From this overview, a representative dataset is identified and interrogated with regards to its representation of the landfill process. Thereafter, the applicability of this dataset to South African landfill conditions is investigated. The investigation focuses on the current capacity of the dataset to represent local landfill practices, identifies factors affecting the landfill inventory, and investigates the parametrisation potential that exists to account for these factors.

5.1 Overview of Landfill Datasets

As discussed in the literature review (Chapter 2), obtaining inventory data for a landfill site is challenging. Primary data collection from landfill sites is unfeasible, due to various factors such as the time frame associated with landfill emissions and the heterogeneity of the waste body. While various landfill emission models have been developed to address the challenges in obtaining material-specific landfill emission data, these are typically not capable of accounting for site specificities influencing waste degradation and emission release. Furthermore, the use of these models can require extensive input data, which can be difficult to obtain. For such purposes as modelling landfill disposal for the end-of-life stage of a product LCA, the use of generic LCA datasets contained within SimaPro provides a practical alternative to developing a landfill inventory by other means.

The following section aims to assess the current capabilities for modelling the landfill disposal of general waste within the context of SimaPro.

5.1.1 Availability of Landfill Datasets within SimaPro

Within SimaPro, inventory data for the landfill process is supplied by two different databases: ecoinvent v3.3 and the US LCI. Within ecoinvent v3.3, three types of landfills are represented: inert material landfills, residual material landfills, and sanitary landfills. Each dataset is dependent on an underlying model, which represents the particular landfill process, and models the emissions associated with that process. For each landfill process, there are a number of different datasets representing the disposal of different waste types. The US LCI database also makes a distinction between landfill type and waste type. However, all of the datasets contained within SimaPro are "dummy" processes, meaning that they are empty datasets. According to the corresponding process record for these dummy datasets within SimaPro, there has been no data collection for these processes and thus it is advised that the user obtain "proxy data from other sources to bridge this data gap" (PRé Consultants, 2016)

Assuming that primary data collection and landfill emission modelling falls beyond the scope of a typical product LCA, landfill inventory data within SimaPro is limited to the datasets contained within the ecoinvent v3.3 database. Therefore, the ecoinvent v3.3 landfill datasets and their representation within SimaPro form the basis of the following discussion.

5.1.2 Overview of Ecoinvent Sanitary Landfill Datasets

As noted above, within ecoinvent v3.3, three types of landfills are represented and for each specific landfill type, various waste types are inventoried. Inert material landfills accept material with a low pollutant content such as excavation material and construction waste, while residual material landfills typically accept industrial waste such as fly ash from incineration processes (Doka, 2003d). Therefore, of the available landfills types, the sanitary landfill process is most representative of landfill disposal of general waste in South Africa. The ecoinvent v3.3 sanitary landfill datasets are based on the operation of well-managed Swiss sanitary landfill operations (Doka, 2003d). According to Doka (2003d), the characteristics of such landfill sites are as follows:

- Base and boundary sealing
- Daily site operations including compaction and covering of waste
- Water collection system
- Gas collection system
- Treatment of collected leachate in a wastewater treatment plant (WWTP)
- Incineration and/or utilisation of LFG
- Restoration and post-closure monitoring of site

The resulting sanitary landfill datasets inventory the infrastructure, resource, and process-specific demands for the landfill process, in addition to the waste-specific emissions from the site. The infrastructure, resources, and other process-specific demands are linked to the sanitary landfill process itself, and thus are independent of waste type and are based on actual site data obtained from representative Swiss landfill sites. The dataset therefore quantifies the particular demand (per kg of waste disposed) for the process i.e. kWh of electricity or litres of diesel required for daily site operations. The burdens, the physical resource uses and emissions resulting from these demands, are introduced into the landfill inventory by linking the demand to a representative ecoinvent dataset containing the inventory associated with the provision of this demand. Waste-specific emissions, by contrast, are generated from a landfill emission model and represent the emissions from the specific waste that is landfilled. Waste-specific emissions include all emissions from the waste incurred from the various treatment processes included within the system boundary for the sanitary landfill process.

Although the ecoinvent v3.3 sanitary landfill datasets are best representative of well-managed Swiss conditions, geographical differences can be introduced to a certain extent through the geographical linking of datasets. This approach allows the specification of country-specific datasets for certain process-specific demands. Through geographic linking of datasets, ecoinvent create three geographical defaults for each sanitary landfill dataset: Switzerland (CH), Europe without Switzerland, and Rest of World (RoW). It should be emphasised that for each geography, the quantified demand is constant, implying that the infrastructure requirements and resource and energy consumption is consistent for a sanitary landfill process, regardless of where it is undertaken. The capability of ecoinvent datasets to incorporate the geographical linking of datasets is supported by their unit process representation within SimaPro, allowing modification to be made to both the quantification of the demand and the linked dataset.

Waste-specific emissions are not linked to an external dataset, as they are generated from the underlying landfill emission model. While the parametrisation of this model allows for the definition of different waste types and/or waste characteristics, the underlying assumptions are based on well-managed Swiss landfill operations. Therefore, although waste type is an adjustable modelling parameter, accounting for the effect of different geographical conditions on landfill emissions is a more complex undertaking, requiring modification of the model. Unlike process-specific demands, modifications to the waste-specific emissions cannot be made within the SimaPro process record. Such modifications require the use of the model to generate a new output, which can then be imported into a process record.

While there is a certain level of adaptability within existing ecoinvent sanitary landfill datasets to represent different geographies and waste type/composition, it is arguable as to whether this is sufficient to represent alternative conditions to those forming the basis of these datasets. In particular, it is questionable as to whether datasets can be adapted to represent landfill disposal under unsanitary or informal conditions. To interrogate the capabilities of these datasets in representing alternative landfill processes, the ecoinvent v3.3 dataset representing the sanitary landfill of municipal waste was chosen for further interrogation. The focus of the investigation was directed towards understanding how the ecoinvent sanitary landfill dataset is constructed and its adaptability, and as such, the choice of waste type has little bearing on the result.

As discussed in the methodology (Chapter 3), the ecoinvent v3.3 dataset was primarily assessed within SimaPro, as this is the medium through which the data is widely accessed by LCA practitioners[11]. Furthermore, SimaPro preserves the modular construction of ecoinvent datasets. This modular construction is a particular benefit of ecoinvent, allowing individual processes contained within the dataset to be represented by separate activity links. This allows the inventory data to be assessed on a unit process level, as opposed to an aggregated system level. Thus, within SimaPro, the unit process record (U) for this dataset was investigated as opposed to the alternative system process record (S). The attributional dataset with recycled content allocation was selected. The SimaPro identifier for this dataset is as follows: Municipal solid waste (RoW)| treatment of, sanitary landfill | Alloc Rec, U.

5.1.3 Representation of the Sanitary Landfill Process in Ecoinvent

A detailed documentation of the development of the ecoinvent sanitary landfill LCI is available in *ecoinvent Report No. 13: Life Cycle Inventories of Waste Treatment Services* (Doka, 2003a), with the most comprehensive information contained in *Part III (Landfills – Underground Deposits – Landfarming)* (Doka, 2003d). It is not the intention of this work to reproduce a detailed discussion on how the sanitary landfill LCI was developed, but rather to provide an overview and analysis of the documented approach with respect to its application to South African landfilling practices and interrogate where adaptation is possible to better represent the South African reality.

A schematic representation of the process chain for the sanitary landfill of general waste, as represented within the ecoinvent v3.3 sanitary landfill dataset, is shown in Figure 5.1 overleaf.

[11] ecoinvent datasets can alternatively be accessed in their primary form through the ecoinvent database. Furthermore, existing datasets can be edited within the freeware “ecoEditor”.

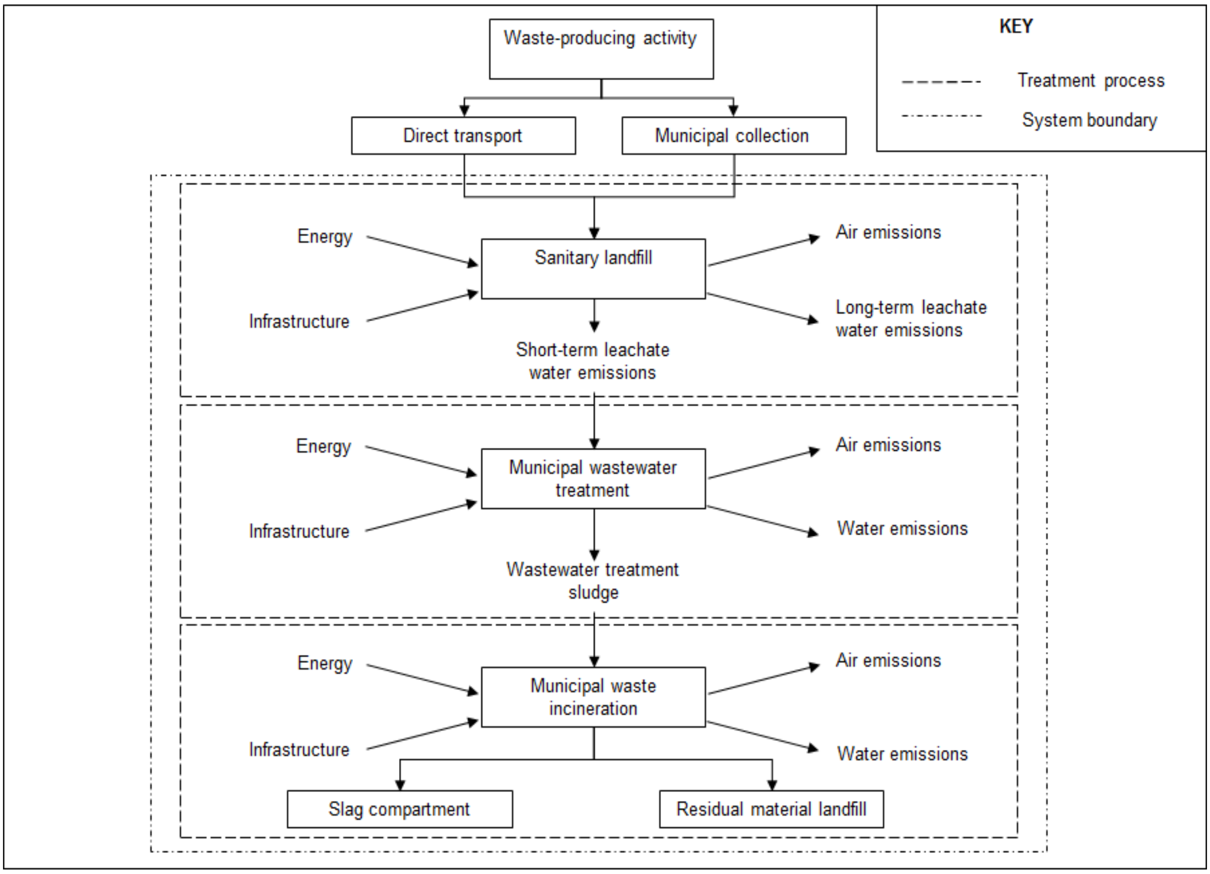

Figure 5.1 Schematic representation of the sanitary landfill process as represented within the ecoinvent v3.3 sanitary landfill inventories (Adapted from Doka, 2003d:38)

As shown in Figure 5.1, the system boundary defined for the sanitary landfill process contains neither the burdens from the waste producing activity nor those associated with the transport of the waste. Within the system boundary, the sanitary landfill process is broken down into three separate processes: the operation of the sanitary landfill site itself, treatment of leachate generated from the waste body in the first 100 years after deposition in a municipal WWTP, and incineration of the resulting wastewater treatment sludge in a municipal incineration plant. The incineration residues are landfilled in slag compartments and residual material landfills (Doka, 2003d). Direct burdens from each process contained within the system boundary include air and water emissions, and land use with indirect burdens arising from energy consumption and infrastructure materials (Doka, 2003d).

Emissions from the landfill site are represented by different data modules, with the distinction between waste-specific and process-specific data modules based on whether or not they depend on the composition of the landfilled waste (Doka, 2003d). The modular construction of the ecoinvent dataset means that within each data module, further differentiation is made between the individual processes (activities) comprising the sanitary landfill process. The benefit of this disaggregation lies in the application and interpretation of the resulting inventory, providing a high level of transparency to the dataset. Given that the system boundary includes the landfill process itself in addition to the leachate treatment and resulting sludge treatment processes (Figure 5.1), linking the emissions to the specific process from which they arise enables the resulting dataset to be assessed on a unit process level. A schematic representation illustrating such a characterisation of the ecoinvent v3.3 sanitary landfill dataset is shown in Figure 5.2 overleaf.

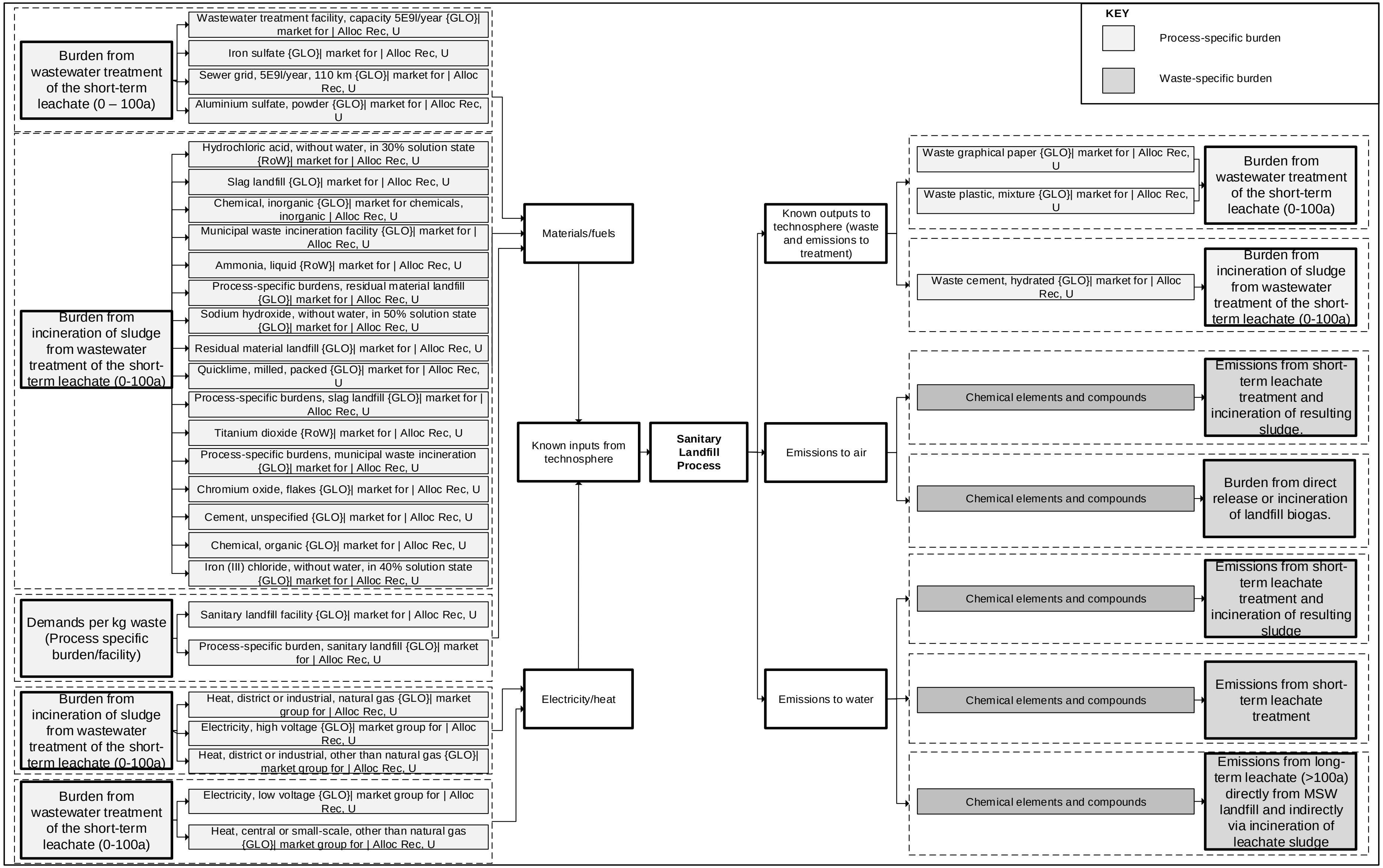

Figure 5.2 Schematic representation illustrating the modular construction of the ecoinvent v3.3 sanitary landfill dataset

Figure 5.2 suggests that the inputs to the sanitary landfill process are largely independent of waste composition and can be grouped according to the process from which they arise i.e. landfill, leachate treatment, or sludge incineration. The outputs from the process are dominated by the emissions from the waste itself and can also be linked to a specific treatment process. This representation allows the input and output to each process to be assessed separately. This has particular benefit when assessing the applicability of the dataset to landfill sites where the treatment processes and technologies might vary from standard Swiss practice.

Given that SimaPro preserves the modular construction of the ecoinvent dataset, within the corresponding SimaPro process record (for the unit process representation of the ecoinvent dataset) the following modifications to the dataset are possible:

- Elimination of irrelevant process and/or waste-specific burdens
- Inclusion of additional process and/or waste-specific burdens
- Redefinition of the quantity of the process and/or waste-specific burdens
- Substitution of alternative datasets to represent process and/or waste-specific burdens
- Modification of existing linked dataset

The modifications outlined above represent a somewhat generic approach to the modification of ecoinvent datasets within SimaPro. While this approach is well suited to modifying process-specific burdens, for waste-specific emissions, this approach is limited and should be undertaken with caution. Waste-specific emissions are determined from an emission model. This has been developed in such a way as to ensure that the principles of mass balance are conserved; the emitted elements have to be consistent with the elemental composition of the waste. Therefore, if waste-specific emissions are modified in the process record, it is necessary to ensure that the modification makes sense in the context of the waste type, and that no more of any element is emitted from the landfill process than was originally present in the waste.

Given that the landfill emission model is representative of standard Swiss landfill practice, the applicability of the output to alternative conditions is questionable. For example, for a site without leachate collection and treatment, emissions to groundwater will differ from those predicted by the model. While a complete modularisation of the waste-specific emissions into process-linked outputs would enable certain generic adaptations to be made to the corresponding process record — such as the removal of emissions from an irrelevant process — the limited disaggregation illustrated in Figure 5.2 prohibits the extent to which generic adaptation can be undertaken. For example, Figure 5.2 shows that for both air and water emissions, the short-term emissions from the leachate treatment and sludge incineration processes are combined and inventoried as one total emission flow. Should it be necessary to represent an alternative sludge management process, these emissions cannot be isolated from the total inventoried flow.

Assuming that primary data collection is unfeasible within the context of a landfill inventory, in order to determine the waste-specific emissions for alternative landfill conditions it is necessary to modify the underlying ecoinvent landfill emission model. The importance of accurately representing waste-specific emissions should not be overlooked, as according to Doka (2003d), the emissions from the waste itself are usually the most important burden within the LCIA result of a landfill. Adaptation of the existing sanitary landfill dataset therefore requires both the relatively straightforward modification of process-specific burdens and the somewhat more complex adaptation of the underlying emission model to determine the waste-specific burdens.

5.2 <u>Applicability of the Ecoinvent Sanitary Landfill Dataset to South African Conditions</u>

This section investigates the extent to which the current ecoinvent v3.3 sanitary landfill dataset represents local landfill conditions, and where it falls short, to what extent adaptation is possible to improve its application to representing landfill disposal in South Africa.

5.2.1 Boundary Definition

As discussed in Section 5.1.3, the system boundary for the ecoinvent v3.3 sanitary landfill dataset incorporates three processes: the sanitary landfill of the waste, treatment of landfill leachate in a WWTP, and incineration of the resulting WWTP sludge. Therefore, the first step in assessing the relevance of this dataset to South Africa is determining whether these processes apply to local landfill operations. Given the range in local landfill operations (as discussed in Chapter 4, Sections 4.6 – 4.7), three generic landfill cases were assumed for South Africa: sanitary landfill, non-engineered landfill, and open dump. It was assumed that basic conditions for each type of site are consistent with those defined in Table 2.6 (Chapter 2). The results of this comparison are shown in Table 5.1.

Table 5.1 Overview of the applicability of treatment processes included within the ecoinvent v3.3 sanitary landfill dataset to generic landfill cases assumed to be reflective of South African landfill practices

Treatment process included in the ecoinvent sanitary landfill dataset	Applicability to South African landfills		
	Sanitary Landfill	Non-Engineered Landfill	Open Dump
Sanitary landfilling of waste	Yes	Partial	Partial
Wastewater treatment of short-term leachate	Yes	No	No
Incineration of sludge from wastewater treatment of short-term leachate	No[a]	No	No

[a] South Africa has virtually no incineration capacity and is dependent on on-site sludge disposal including direct land application and stockpiling of sludge on site (Snyman, 2010).

Table 5.1 provides a useful overview with regards to the relevance of the processes included in the ecoinvent v3.3 sanitary landfill dataset to South African landfill practices. Sanitary landfill sites in South Africa typically reflect a certain level of landfill infrastructure, aimed at containing and managing emissions from the waste body. Hence, both the waste itself and subsequent emissions undergo some form of treatment. According to the *Minimum Requirements for Waste Disposal by Landfill* (DWAF, 1998c) in South Africa, operating and infrastructure requirements become increasingly stringent for larger landfills (see Table 4.9 and Table 4.10). This implies that the existing ecoinvent sanitary landfill dataset is likely to be better representative of larger, sanitary landfill conditions than smaller, unmanaged landfills, and open dumps.

Given the reduced controls in place for non-engineered landfill and open dumps, the representation of treatment processes such as wastewater treatment and sludge management are potentially irrelevant for such sites. However, for both cases the burdens associated with the landfilling process itself cannot be excluded. For non-engineered landfills, a certain level of infrastructure can exist, with evidence suggesting that covering and compaction does occur to a certain extent. While an open dump has limited processing or infrastructure burdens, the emissions associated with waste degradation and land use must be accounted for. With regards to the latter, while the ecoinvent v3.3 sanitary landfill dataset does account for land use burdens, it can be surmised that these are representative of large, sanitary landfill operation, and hence underestimate non-engineered operations and open dumps. However, land use burdens are not a focus of this work.

5.2.2 Site Conditions Affecting Landfill Emissions

Given the climatic variation occurring across South Africa, in terms of modelling the impacts associated with landfill disposal, it is potentially relevant to consider the leachate generation potential of different sites. According to the results presented in Chapter 4 (Section 4.6), approximately half of all formal large landfill sites in South Africa have the potential for significant leachate generation (B^+). However, for medium, small, and communal size classes, the majority of sites (70%, 84%, and 90%, respectively) are classified as B^-. The B^- site classification denotes that less-stringent leachate monitoring and control systems are required due to the reduced leachate generation potential — and hence environmental impact — of such sites.

In addition to variation in leachate generation and management, South African landfill sites further exhibit variation with regards to LFG capture and management. Although the *Minimum Requirements for Waste Disposal by Landfill* (DWAF, 1998c) (Table 4.9 and Table 4.10) stipulate that medium and large landfill sites must have a LFG management system in place, currently passive venting of LFG to the atmosphere is the most common form of gas management, with very few gas collection and utilisation systems in operation in South Africa (Bogner & Lee, 2005, Friedrich & Trois, 2013b).

Both LFG and leachate emissions can have a significant influence on the potential impacts of a landfill site. Therefore, factors affecting the generation, collection, and treatment of these emissions are important to consider in landfill modelling. A comparison of such factors for the ecoinvent sanitary landfill conditions and average South African sanitary landfill conditions is shown in Table 5.2 overleaf.

According to Table 5.2, South African sanitary landfills have comparably lower LFG collection and utilisation capacity than standard Swiss conditions. In South Africa, the majority of LFG is released into the atmosphere without any treatment. Given that the ecoinvent sanitary landfill dataset assumes that 47% of LFG is captured and either flared or utilised for energy recovery, the use thereof will likely underestimate the LFG emissions and subsequent potential impacts of South African landfill sites. In terms of leachate emissions, a major discrepancy between the ecoinvent and South African conditions occur in the management of WWTP sludge. Unlike Switzerland, South Africa has limited incineration capacity, and hence depends on on-site disposal methods such as direct land application and stockpiling of sludge on site (Snyman, 2010). Given that the majority of South African landfill sites have a B^- classification, the question arises as to whether South African landfill sights generate sufficient leachate for this discrepancy to have a notable effect on the potential impacts of the site.

The comparison shown in Table 5.2 does not account for the possibility of non-compliance to regulated conditions, nor does it consider unsanitary or non-engineered sites. In terms of LFG generation, the waste degradation environment affects the quantity of CH_4 that is formed. Under anaerobic conditions (as assumed in the ecoinvent v3.3 sanitary landfill dataset), organic matter decomposes to produce CH_4, whereas under aerobic conditions, CO_2 is predominantly produced (IPCC, 2006b). As discussed in Section 2.5.3 (Chapter 2), the waste degradation environment can be affected by a number of factors including inadequate covering and compaction, human and animal scavenging on the operational face of the landfill, and open burning. To account for the effect these conditions can have on the degradation environment of the landfill site, and hence the LFG emissions which occur, the use of a MCF has been proposed by the IPCC (2006b) (see Table 2.9, Chapter 2). Although the *Minimum Requirements for Waste Disposal by Landfill* (DWAF, 1998c) are intended to prohibit such conditions from occurring on medium and large licensed landfill sites, as highlighted in Section 4.7 (Chapter 4), non-compliance can occur with appreciable frequency on South African sites.

Table 5.2 Comparison of the sanitary landfill conditions assumed in the ecoinvent v3.3 sanitary landfill dataset to average South African sanitary landfill conditions

Landfill Characteristics	Ecoinvent Sanitary Landfill Dataset[a]	South African Sanitary Landfill	
B^+ = Significant leachate generation potential B^- = Sporadic leachate generation		**B^+**	**B^-**
Short-term air emissions			
Waste degradation environment	Anaerobic MCF = 1	Anaerobic MCF = 1	
LFG capture infrastructure	Requirement for all sites	South Africa has limited LFG collection infrastructure and depends on passive venting[b,c] Five registered Clean Development Mechanism (CDM) LFG extraction and utilisation projects (consisting of ten landfill sites)[c]	
LFG capture efficiency	47%	Subject to high variability. Data from CDM project operating in eThekwini Municipality showed instantaneous collection efficiencies of 75% could be reached in the local context[b]	
Captured landfill gas to flare	34%	Of the ten landfill sites that have implemented LFG extraction and utilisation projects, only eight are still operational with two successfully generating electricity[c]	
Captured landfill gas to utilisation	66%		
Short-term leachate (0 – 100 years)			
Capture efficiency	100%	100%	n/a
Leachate treatment	WWTP	WWTP or on-site storage and treatment	n/a
Leachate sludge management	Incineration with disposal in slag landfill and residual material landfill	On-site disposal including direct land application and stockpiling of sludge on site[d]	n/a
Long-term leachate (> 100 years)			
Leachate capture	Base layer failure: all emissions to groundwater	Base layer failure: all emissions to groundwater	n/a

[a] Ecoinvent dataset conditions as reported by Doka (2003d)
[b] As reported by Friedrich & Trois (2013b)
[c] As reported by Bhailall (2015)
[d] As reported by Snyman (2010)

Non-compliance to regulated conditions and/or lack of engineering control can also have an effect on the leachate emissions from a landfill site. For medium and large B^+ sites, the *Minimum Requirements for Waste Disposal by Landfill* (DWAF, 1998c) stipulate that all sites need to be lined and have a leachate detection, management, and treatment system in place (see Table 4.9 and Table 4.10). For small B^+ sites, these requirements are reduced but still require some form of liner and leachate management system. The audit report analysis presented in Chapter 4 (Section 4.7) indicates that non-compliance to leachate management requirements occurs with an appreciable frequency in South Africa. Furthermore, a number of unregulated sites are in operation. This could result in the direct release of short-term leachate emissions into the soil or groundwater beneath the site. Additionally, due to South Africa's historical dependence on landfill, a number of licensed South African sites lack even basic leachate containment systems, such as a base liner, as they were operational before the publication of any formal landfill requirements (Käsner, personal communication 2016, 15 May 2016). The potential for the direct release of untreated landfill leachate implies that the impacts of leachate emissions from South African B^+ landfill sites could be higher than those incurred with the use of the ecoinvent v3.3 sanitary landfill dataset.

Although B^- sites do not require any leachate management, containment, or treatment infrastructure (as it is assumed their sporadic leachate generation potential will not result in sufficient leachate to warrant a leachate management system (DWAF, 1998c)), this classification can be misleading in accounting for leachate emissions from South African landfill sites. The differentiation between B^+ and B^- sites is based on a simplified climatic water balance, which takes into account only rainfall and evaporation (DWAF, 1998c:s3, ss3.4). It has been acknowledged that a number of non-climatic factors could affect the water balance on a site such as poor siting or infrastructure design and inadequate maintenance and site operations (DWAF, 1998c:s3, ss3.4). Should these factors affect the climatic water balance such that a B^- site generates significant leachate, this will result in the uncontrolled release of leachate, which has implications for the environmental impacts of the site. Under such circumstances, the *Minimum Requirements for Waste Disposal by Landfill* (DWAF, 1998c:s3, ss3.4) stipulate that the site be reclassified and remedial action taken to mitigate the impact of the emitted leachate .

While uncontrolled leachate emissions can occur from both B^+ and B^- sites, quantifying this potential is challenging, particularly with regards to B^- sites, which do not require leachate monitoring systems. Obtaining a quantified leachate estimate is further complicated by the continued operation of unregulated or unlicensed sites, for which limited information regarding leachate generation is available. Without detailed accounting of the relative volumes of leachate generated from different landfill sites (including uncontrolled releases and sporadic leachate generation from B^- sites), it is difficult to quantify the potential for uncontrolled leachate release in a representative landfill dataset for South Africa.

5.2.3 Process-Specific Burdens

As discussed in Section 5.1.3, generic modifications that can be made in SimaPro to the process-specific burdens in the ecoinvent v3.3 sanitary landfill dataset include removing or substituting the linked dataset representing a specific burden, and/or updating the value of the burden to reflect South African conditions. For processes included in the dataset that are irrelevant to landfill practices in South Africa, the associated burdens can simply be removed. Conversely, where an alternative process is undertaken on South African sites, the resulting burdens need to be incorporated into the inventory.

Following the research undertaken by Friedrich and Trois (2013b), South African landfill operations are relatively well documented with regards to process-specific demands contributing towards GHG emissions. However, the reported burdens are not exhaustive in accounting for all process-specific demands associated with the process, notably excluding any burdens associated with downstream leachate treatment and sludge disposal. An overview of the process-specific burdens included in the ecoinvent v3.3 sanitary landfill dataset and their potential for modification based on the availability of representative South African input values and regional datasets is shown in Table 5.3 overleaf. Given South Africa’s limited incineration capacity, process-specific burdens associated with this process are excluded from further analysis and are not shown in Table 5.3. It should be noted that each process-specific burden included in the sanitary landfill inventory is linked to a representative dataset. Therefore, each linked dataset can be further analysed with regards to its capabilities in representing South African conditions. This exercise was not undertaken, with the exception of the process-specific burdens for the sanitary landfill site, where the linked datasets for “known inputs from technosphere” were interrogated in further detail. These are shown in Table 5.4 (pg. 96).

Table 5.3 Overview of the process-specific burdens included in the ecoinvent v3.3 sanitary landfill dataset and their potential for modification based on the availability of representative South African input values and regional datasets

Description of Burden[a]	Demand per kg wet MSW		South African Regional Dataset?
	Inventory Default Demand[b]	South African Demand	
Sanitary landfill facility (CH) | market for | Alloc Rec, U			
Infrastructure materials for landfill construction, operation and aftercare of 1.8 million m^3 landfill for untreated municipal waste. Geography specific to the technology encountered in Switzerland in 2000. Well applicable to modern landfilling practices in Europe, North America, or Japan	5.56E-10 p[c]	Data required	No
Process-specific burden, sanitary landfill (CH) | processing | Alloc Rec, U			
Process-specific energy demand and land use of landfill. Geography specific to the technology encountered in Switzerland in 2000. Well applicable to modern landfilling practices in Europe, North America, or Japan	1 kg[d]		No
Electricity, low voltage (CH)| market for | Alloc Rec, U			
Electricity demand for wastewater treatment of short-term leachate	0.00864 kWh	Data required	Yes
Heat, central or small-scale, other than natural gas (CH) | market for | Alloc Rec, U			
Heat for heating the digester and general space heating	0.00054238 MJ	Data required	No
Wastewater treatment facility, capacity 5E9l/year (CH)| market for wastewater treatment facility, capacity 5E9l/year | Alloc Rec, U			
For municipal wastewater treatment plant capacity class 3 (average plant size in Switzerland) a lifetime of 30 years is assumed. Infrastructure materials for municipal wastewater treatment plant, transports, dismantling. Specific to the technology mix encountered in Switzerland in 2000. Well applicable to modern treatment practices in Europe, North America or Japan.	1.42E-11 p	Data required	No
Iron sulfate (GLO)| market for | Alloc Rec, U			
Agent for precipitation of dissolved phosphate in WWTP	6.43E-5 kg	Data required	No
Sewer grid, 5E9l/year, 110 km (CH)| market for sewer grid, 5E9l/year, 110 km | Alloc Rec, U			
Infrastructure materials for municipal sewer system, transports, dismantling. Specific to the technology mix and human settlement structure encountered in Switzerland around 1996.	5.45E-10 km	Data required	No
Aluminium sulfate, powder (GLO)| market for | Alloc Rec, U			
Agent for precipitation of dissolved phosphate in WWTP	1.74E-5 kg	Data required	No

[a] Adapted from the documentation provided in the SimaPro representation of the ecoinvent sanitary landfill dataset and/or Doka (2003d) and (Doka, 2003b).
[b] As reported in Doka (2003d) and Doka (2003b).
[c] Demand per kilogram of waste. Reported as a fraction of the demand associated with the construction of a 1.8 million m^3 landfill for untreated municipal waste.
[d] Process-specific burdens for the sanitary landfill operation determined on the basis of treatment of 1 kg waste. Dataset contains links to individual burdens.

Table 5.4 Overview of the linked datasets contained in the "known inputs from technosphere" category for the ecoinvent process-specific burden, sanitary landfill (CH) | processing | Alloc Rec, U dataset and their potential for modification based on the availability of representative South African input values and regional datasets

Description of Burden[a]	**Demand per kg wet MSW**		**South African Regional Dataset?**
	Inventory Default Demand[b]	**South African Demand**	
Diesel, burned in building machine (GLO)| market for | Alloc Rec, U			
Diesel consumption of special loaders used to distribute and compact the waste. Certain elementary flows or intermediate product flows are extrapolated from European and Swiss conditions	0.0467 MJ (1.3 litres/tonne waste)	0.34 litres/tonne waste[c]	No
Heat, central or small-scale, other than natural gas (CH)| market for | Alloc Rec, U			
Energy demand of simple administrative house	0.0015134 MJ	n/a	No
Electricity, medium voltage (CH)| market for | Alloc Rec, U			
Operation of landfill gas pumps	0.00135 kWh	0.009 kWh[d]	Yes
Electricity, low voltage (CH)| market for | Alloc Rec, U			
Energy demand of simple administrative house	1.5E-5 kWh	0.009 kWh[d]	Yes

[a] Adapted from the documentation provided in the SimaPro representation of the ecoinvent sanitary landfill dataset and/or Doka (2003d) and (Doka, 2003b).
[b] As reported in Doka (2003d) and (Doka, 2003b)
[c] Reported as diesel for daily on-site operations by Friedrich & Trois (2013b)
[d] Value reported for "Electricity for on-site lighting, administration buildings, pumps and fans" by Friedrich & Trois (2013b) with no distinction made between medium and high voltage demand.

The results of Table 5.3 and Table 5.4 suggest that there is a limitation in the availability of data for inventorying the process-specific demands of South African landfill operations, particularly with regards to landfill infrastructure and downstream leachate treatment. If it is considered that only medium and large sanitary landfill sites with a positive climatic water balance (B^+) require both comprehensive leachate management and treatment systems, a more pertinent question is whether these burdens are relevant to landfilling in South Africa. Considering the results presented in analysis Table 4.8 (Chapter 4), approximately 50% of large landfill sites are classified as B^+, while for medium sites, this decreases to 30%. Using the estimated waste distribution between landfill classes (Table 4.8), this implies that less than 40% of formally managed waste is treated in landfills where leachate capture and treatment is necessary. If informal waste disposal is considered, the relative proportion of waste generating leachate that is treated in WWTP or similar, will decrease further.

Considering the observations of Doka (2003d) and the discussion presented in Section 2.4.2 (Chapter 2), the impacts associated with the landfill process are typically dominated by the emissions from the waste itself. Thus, the relatively low impacts from process-specific burdens combined with the observation that leachate treatment is applicable to less than 40% of landfilled waste suggests that obtaining data for these missing burdens should not be regarded as a priority for inventorying landfill emissions in South Africa.

5.2.4 Waste-Specific Burdens

Direct emissions from the waste body are emitted through two media, air and water, as LFG and leachate, respectively. These emissions are primarily dependent on the waste composition, waste reactivity and its degradability (Doka, 2003d) and represent the burdens associated with the waste itself i.e. waste-specific burdens. As illustrated by Figure 5.2, the modular construction of the ecoinvent dataset allows the waste-specific emissions to be inventoried separately, allowing the emissions to be linked to the individual processes from which they arise. Within the dataset, further distinction is made with regards to the time frame for these emissions and the compartment and sub-compartment for their release. The categorisation of emissions in particular is important for quantifying their relative impacts in the LCIA stage.

As noted, the waste-specific emissions inventoried in the ecoinvent sanitary landfill dataset are dependent on the output from the underlying ecoinvent landfill emission model. However, a short-coming in the use of inventory models to determine landfill emissions is that they do not consistently account for the effect that variations in landfill specificities (such as climate, rainfall, height, cover type, waste density, and permeability) have on waste decomposition and emission release (Obersteiner et al., 2007). Given that the ecoinvent landfill emission model is based on Swiss landfill conditions — and limited alternative datasets are available in SimaPro for alternative landfill processes such as unsanitary landfill sites and open dumps — to account for the influence of geography and waste management practices on waste-specific emissions, it is necessary to modify the underlying model. The modified model can then, theoretically, be used to generate waste-specific emissions for South African conditions.

It should, however, be considered that in reality, few users would want to use this model at the level considered in this analysis. Furthermore, it is not the intention of the model to be capable of representing alternative landfill practices. If it is desired to develop an inventory for alternative landfill practices, given the scope defined for this research, it is necessary to consider these potential modifications in terms of those which can be practically implemented by a product designer or LCA practitioner. Assuming that detailed primary data collection and landfill modelling falls outside of the scope of a typical product LCA, it is necessary to make a distinction between what can be considered practical and impractical in terms of adaptation potential.

A necessary first step in assessing the modification potential of the ecoinvent sanitary landfill model is understanding the model and evaluating its current capabilities with regards to representing South African landfill conditions. As previously noted, detailed documentation of the development of the ecoinvent sanitary landfill model is available in *ecoinvent Report No. 13: Life Cycle Inventories of Waste Treatment Services Part III (Landfills – Underground Deposits – Landfarming)* (Doka, 2003d). Thus, the following sub-sections will not provide a detailed discussion on the model development, but rather focus on providing an overview of how waste-specific emissions are determined. This model is based on the established concept of using waste composition data and transfer coefficients for different chemical elements to calculate waste-specific emissions (Doka, 2003d). The novelty of this model lies in its determination of waste-specific degradability, incorporation of a release factor to account for the effects of reprecipitation, the consideration of preferential flow of leachate transport, and approach for modelling long-term emissions (Doka, 2003d).

5.2.4.1 Emissions from the Waste Body

Modelling direct emissions from a landfill site poses a number of challenges due, amongst else, to the heterogeneity of waste material and the time frame associated with waste degradation and emission release. With regards to the latter, given the potential for landfills to continue to emit pollutants for centuries beyond waste deposition (Doka, 2003d), both short- (0 – 100 years) and long-term (100 – 60 000 years) emissions are accounted for by the ecoinvent emission model. However, due to the somewhat "controversial perception" of including future impacts in LCA (Doka, 2003d:17), the resulting dataset makes a rudimentary temporal distinction: short- and long-term emissions are inventoried separately, but are assigned the same impact factor (Doka, 2003d). The benefit of the separation is seen in the LCIA stage of the LCA, where the total landfill impacts can be disaggregated and assessed from both a long- and short-term perspective.

Direct emissions from the waste body are waste-specific and determined for individual elements (e). Hence, the model is dependent on the properties of the deposited waste, such as its chemical composition and the transfer coefficient (TK) of the element, which determines the quantity of the element emitted from the disposed material. This relationship underpins the ecoinvent landfill emission model and is shown in Equation 5.1 (as defined by Doka (2003d:19)).

$$\text{Emission}_{\text{media, phase, e}}\left[\frac{kg_e}{kg_{waste}}\right] = TK_{\text{media,phase,e}} \times \text{waste composition}_e\left[\frac{kg_e}{kg_{waste}}\right]$$

Equation 5.1

TKs are determined to describe the movement of an element in either landfill gas or leachate (media) for both the short- and long-term phase of the landfill operation. In the general case, the TK represents the average behaviour of an element, and is calculated from average operation and average waste composition (Doka, 2003d). However, in the case of the ecoinvent sanitary landfill model, TKs are modified according to the degradability of the specific waste type, thus allowing the determination of waste-specific emissions (Doka, 2003d).

The degradability of the waste is a key variable underpinning the waste-specific emissions model. Waste degradability can be defined as the "decomposition and mineralisation of the materials in a waste matrix" (Doka, 2003d:42). Degradability is represented as a percentage, ranging from 0% (no decomposition within the first 100 years after deposition) to 100% (complete degradability within 100 years), with intermediate degradability falling between the two extremes (Doka, 2003d). For waste with 0% degradability, no emissions occur during the methane phase of the landfill, and all waste-specific pollutants are emitted as long-term emissions only. For fully degradable waste, the waste matrix is completely destroyed within 100 years after deposition (Doka, 2003d).

The extent of degradability cannot be used exclusively as a measure of the emissions occurring from the landfill body due to the effects of re-precipitation. These effects are seen in storage deposits of pollutants and delayed emissions from the waste body (Doka, 2003d). Thus, according to Doka (2003d:42) for waste with an intermediate degradability, after 100 years the pollutants contained within the waste will have one of four fates:

1. Not-decomposed
2. Decomposed but re-precipitated and not released
3. Decomposed and released in landfill leachate
4. Decomposed and released in landfill gas

Both the degradability of the waste and the release factor represent important parameters in the determination of the emission output from a landfill site. For short-term emissions, an additional parameter is required to distinguish between leachate and gas emissions. This gas release factor expresses the proportion of the emissions that occur as gas emissions in the first 100 years (Doka, 2003d). Using this factor, the short-term emissions from the landfill site are determined from Equation 5.2 and Equation 5.3 (as defined by Doka (2003d:48)).

$$\text{Emission}_{\text{short-term,gas,e}} = m_e.D.r_e.\text{gas\%}_e$$

Equation 5.2

$$\text{Emission}_{\text{short-term,leach,e}} = m_e.D.r_e.(1\text{-gas\%}_e)$$

Equation 5.3

Where:

$E_{short\text{-}term,gas,e}$ = Short-term emission of the element *e* to landfill gas [kg/kg waste]

$E_{short\text{-}term,leach,e}$ = Short-term emission of the element *e* to leachate [kg/kg waste]

m_e = Concentration of element *e* in waste fraction [kg/kg waste]

D = Decomposition rate of waste [kg/kg in 100 years]

r_e = Average release factor for element *e* [kg/kg]

$\%gas_e$ = Fraction of the released amount of element *e* emitted in landfill gas [weight %]

Comparison of Equation 5.2 and Equation 5.3 with Equation 5.1 shows that the short-term waste-specific TKs can be defined as follows (as according to Doka (2003d:48-49)).

$$TK_{short\text{-}term,\ gas,e} = D.r_e.gas\%_e$$

Equation 5.4

$$TK_{short\text{-}term,\ leach,e} = D.r_e.(1\text{-}gas\%_e)$$

Equation 5.5

Where:

$TK_{short\text{-}term,gas,e}$ = Short-term transfer coefficient of the element e to landfill gas [kg/kg] element

$TK_{short\text{-}term,leach,e}$ = Short-term transfer coefficient of the element e to leachate [kg/kg] element

While the TK enables the determination of emissions from the waste body, these do not necessarily equate to releases to the environment. Within the ecoinvent sanitary landfill model, it is assumed that 47% of the LFG is collected and incinerated, while 100% the short-term leachate emissions are collected and treated in a municipal WWTP (Doka, 2003d). Thus, the emissions from the waste body require further modelling before releases to the environment can be inventoried.

As noted, the short-term emissions represent the pollutants emitted from the waste body 0 – 100 years after the deposition of waste. During this time, not all waste types undergo complete degradation, neither are all released pollutants necessarily emitted, thus it is necessary to model the long-term emissions from the landfill body. Long-term emissions occur after the methane phase of the waste degradation and thus, emissions to air are assumed to be negligible, with all remaining pollutants being emitted via leachate (Doka, 2003d). Given the long time frame considered for these emissions and the variation in conditions likely to occur over this timeframe, modelling these emissions is more complex than the approach required for short-term emissions. A detailed discussion of the approach required to develop a set of waste-specific long-term TKs is available in Doka (2003d).

5.2.4.2 Leachate Emissions

As shown in Figure 5.1, the system boundary for the ecoinvent sanitary landfill dataset incorporates the treatment of leachate generated within the first 100 years after waste deposition in a municipal WWTP and incineration of the resulting wastewater treatment sludge. Therefore, the leachate emitted from the waste body undergoes further treatment — hence requiring further modelling — before releases to the environment can be determined. Within the ecoinvent database, wastewater treatment inventories are generated from a separate WWTP model. The system boundaries for this model include the canalisation of wastewater, the WWTP itself, and the disposal of digester sludge (Doka, 2003b).

In order to generate an inventory for the waste-specific emissions from the sanitary landfill process as a whole — as opposed to developing a separate inventory for each constituent treatment processes — the WWTP model is incorporated into the sanitary landfill model. This is achieved with the use of “burden factors” — individual factors describing the burdens associated with 1 kg of pollutant in 1 m^3 of wastewater (Doka, 2003d:54). These burden factors contain both the direct burdens from the wastewater treatment process itself and the downstream burdens from sludge disposal (Doka, 2003d). For each pollutant in the leachate, a list of burden factors can be developed, for each of the different “exchanges”[12] occurring during the treatment process (Doka, 2003d:54). The burdens from the wastewater treatment process are fully additive and include both the burdens caused by the pollutant themselves in addition to the “base burdens”, which occur irrespective of the pollutant content and represent the burdens associated with the “carrier water” that passes through the WWTP (Doka, 2003d:55). The leachate treatment burdens associated with a particular pollutant, x, and used in the sanitary landfill model is shown in Equation 5.6, as defined by Doka (2003d:55).

$$B_{i,x}=m_x \times (B^0_{i,x}-B_{i,W})$$

Equation 5.6

Where:

$B_{i,x}$ = Burden for exchange *i* from pollutant *x* in leachate

m_x = Mass of pollutant *x* in leachate 0 – 100 years [kg]

$B^0_{i,x}$ = Burden for exchange *i* from 1 kg pollutant *x* in 1 m^3 of wastewater (from WWTP model)

$B_{i,W}$ = Burden for exchange *i* from 1 m^3 of unpolluted wastewater (from WWTP model)

The total burdens associated with the leachate treatment process are calculated from the sum of all of the burdens associated with each pollutant (Equation 5.6) and the base burdens, created by the volume of leachate generated over 100 years and treated in the WWTP (Doka, 2003d). The total burden for leachate treatment used in the sanitary landfill model as defined by Doka (2003d:55) is shown in Equation 5.7.

$$B_i=(B_{i,w} \times V \times 100 \text{ years})+\sum B_{i,x}$$

Equation 5.7

Where:

B_i = Total burden for exchange *i* from leachate treatment

V = Mean annual leachate output from the landfill per kg of waste [m^3/year per kg waste]

Although the burden factor is useful in linking leachate generation to the burdens associated with its treatment, such a representation limits the adaptability of the sanitary landfill model with regards to representing alternative leachate management scenarios. The burden factors are representative of

[12] “Exchanges” refer to the emissions associated with the WWTP itself and those incurred from associated processes and materials (Doka, 2003d).

specific conditions defined in the WWTP model (which, like the sanitary landfill model is based on well-managed Swiss conditions[13]) and cannot be disaggregated into the separate processes comprising the model. A single factor therefore represents all the burdens associated with the wastewater treatment process and sludge incineration. While it is likely that the WWTP is comparable to South African practice, the burdens associated with sludge incineration are not reflective of sludge disposal undertaken in South Africa. However, given that less than 40% of the domestic waste generated in South Africa is disposed of into landfill sites where leachate treatment is undertaken, correcting this burden factor to better reflect sludge management is potentially of lesser importance than modelling the impacts of uncontrolled leachate releases.

5.3 Parameterisation Potential of the Ecoinvent Landfill Emission Model

The focus of the previous section was directed towards understanding how waste-specific emissions are determined within the ecoinvent landfill emission model and their applicability to South African landfill conditions. This section provides an overview of the factors influencing the model output and the parametrisation potential of the model to represent alternative landfill conditions.

5.3.1 Infrastructure, Engineering Controls, and Site Operation

By definition, sanitary landfill sites are controlled operations with various barrier and containment systems in place to manage the emissions from the site. By contrast, non-engineered landfill sites and open dumps typically lack the infrastructure and controls required to adequately manage or contain emissions from the waste body. It can be inferred that the physical characteristics of different landfill sites will influence both emission generation and release. The extent to which the current ecoinvent landfill emission model can be modified to take into account the effect of infrastructure and operating controls on LFG and leachate emissions is discussed in the following sub-sections.

5.3.1.1 Effect of Infrastructure on Landfill Gas Emissions

During the CH_4 generating phase of the waste decomposition, LFG can be collected and is typically incinerated in open flares, furnaces, or gas motors (Doka, 2003d). Depending on which technology is used, energy can be recovered in the form of electricity or useful heat (Doka, 2003d). Within well-managed sanitary landfill sites, the proportion of landfill gas that is captured can vary, but is typically < 60% (Doka, 2016). Within the current ecoinvent sanitary landfill model, a capture efficiency of 47% is modelled, of which 34% is flared (with no energy recovery) and the remaining 66% is utilised for energy (Doka, 2003d). For the utilised LFG, both thermal and electrical efficiencies are considered, and are based on the weighted efficiencies of converting motors and furnaces installed in Swiss landfills, reported at 13.5% and 27.8% for thermal and electrical efficiency, respectively (Doka, 2003d).

LFG capture and utilisation is well parametrised within the ecoinvent sanitary landfill model, allowing the specification of appropriate inputs for the percentage of gas captured, the proportion thereof that is either flared or utilised, and the relative efficiency of the technology mix used for electricity or heat recovery. In terms of the efficiencies, it should be noted that the model uses a weighted average efficiency, which takes into account the proportion of gas utilised for each energy output. Modifying these efficiencies therefore requires that the weighted efficiency for converting motors and furnaces in an alternative geographical context be known.

[13] The WWTP includes a three-stage treatment of waste water and digestion of raw treatment sludge as according to the technology mix in Switzerland (Doka, 2003b).

If it is assumed that LFG generation is comparable regardless of the infrastructure and operations occurring on the site, then the existing parametrisation of the model is capable of representing the emission output from well-managed sanitary sites with a high LFG capture efficiency, and non-engineered/unsanitary sites with little or no LFG capture. Furthermore, if it assumed that Swiss technology is comparable to South African technology in terms of energy/heat recovery and operating efficiencies, then adapting the model for South African conditions is relatively straightforward, requiring the specification of only the percentage gas captured and the relative proportion that is flared versus that utilised for energy or heat recovery. Correcting the model to account for alternative LFG utilisation technologies and efficiencies would be a more complex undertaking. However, of the ten landfill sites that have implemented LFG extraction and utilisation projects in South Africa, only eight are still operational with two successfully generating electricity (Bhailall, 2015). These projects were established as part of the CDM programme of the Kyoto Protocol and hence are likely to reflect current technology. This implies that the model default efficiencies are likely to be applicable to South African conditions.

5.3.1.2 Effect of Infrastructure on Leachate Emissions

Within the ecoinvent sanitary landfill model, it is assumed that for the first 100 years after waste deposition, all leachate emissions generated from the waste body are not emitted directly but rather are collected, discharged to a sewer, and treated in a municipal WWTP (Doka, 2003d). After 100 years, it is assumed that the barrier system fails and leachate emissions are emitted to the groundwater (Doka, 2003d). While the long-term emissions from a landfill site are therefore largely independent of any leachate control measures imposed on the site, short-term leachate emissions are directly affected by the level of control, such as landfill liner or leachate collection systems. Therefore, for sites with limited leachate management systems, modification of the model is necessary such that leachate is inventoried as a direct emission to the biosphere as opposed to undergoing further treatment in a WWTP.

Short-term leachate emissions are not parametrised within the ecoinvent landfill model. Given that leachate is an important emission from landfill sites in terms of environmental impacts, this can be considered a shortcoming in the adaptability of the model to represent alternative landfill conditions. Using the existing model structure it is possible to implement a rudimentary modification to approximate the conditions of an unmanaged landfill or open dump. Within the current model structure, short-term leachate emissions from the waste body (kg/kg waste) are generated using Equation 5.1 – Equation 5.5 (Section 5.2.4). These emissions are used as an input to the WWTP, and hence undergo further modelling before the final leachate emissions are inventoried. Therefore, isolating the short-term leachate emissions *before* they undergo further treatment and specifying a relevant sub-compartment (i.e. groundwater, river etc.) provides a proxy for direct leachate emissions from unmanaged landfill sites. Although the assumption that all short-term leachate can be modelled as a direct emission to groundwater represents a simplification of a complex hydrological system, this approach provides an indication of the potential impacts associated with unmanaged leachate. No modification is necessary for the long-term emissions, given that the existing model assumes a system failure after 100 years and hence, all long-term emissions as inventoried as direct emissions to groundwater.

While the approach to approximate uncontrolled leachate release outlined above is highly simplified, alternative approaches in which the model itself is modified to include parameters such as capture and containment efficiencies, wastewater treatment processes and sludge disposal are constrained by the model structure. As discussed in Section 5.2.4, for short-term leachate emissions, the current model is dependent on burden factors derived from the independent WWTP model. Each burden factor represents the burdens associated with wastewater treatment and sludge disposal and cannot be disaggregated into the constituent processes. Modification thereof requires adaption of the WWTP model from which these factors were derived. Based on the latter modification, new burden factors could theoretically be determined and imported into the sanitary landfill model. However, there are major limitations to this approach. Although the WWTP model allows the user to specify certain parameters, such as the size class of the treatment plant and the sludge disposal process, this specification is limited to predetermined options, which are informed by typical Swiss practice. With regards to sludge disposal, the parameterisation is limited to incineration and agricultural application. The exclusion of landfill as a disposal option for wastewater sludge was justified on the basis that landfilling of sludge is not legal in Switzerland and therefore is not considered a viable disposal option (Doka, 2003b).

The exclusion of landfill as a sludge management option limits the applicability of this dataset to the rest of the world, especially developing countries — such as South Africa — with a strong dependence on landfill and limited incineration capacity. Therefore, a burden factor determined from the existing WWTP model reflecting the burdens associated with the incineration/agricultural application of wastewater sludge is somewhat inaccurate within the context of local landfill operations. Accurately representing the treatment of leachate from the waste body in a South African context would therefore require that alternative sludge management practices be incorporated into the WWTP process model from which relevant burden factors can then be derived.

However, both the feasibility of this undertaking and the necessity of representing South African sludge disposal practices are questionable. As previously discussed, given that less than 40% of domestic waste is estimated to be disposed of in landfill sites with leachate collection and treatment infrastructure, the relative impacts incurred by the wastewater sludge disposal process are unlikely to be significant within the broader context of landfill impacts in South Africa. Given that on average, the leachate generation potential of South African sites is likely to be lower than their Swiss counterparts, due to the climatic differences between South Africa and Switzerland, it is likely that a more pertinent issue is correcting the model to account for the relative volume of leachate that is generated under South African conditions. The effect of climate on leachate emissions and the parametrisation potential of the model with regards to accounting for these factors are discussed in further detail in Section 5.3.2.

5.3.1.3 Effect of Site Operations on Emission Generation

The ecoinvent sanitary landfill emission model assumes that waste degradation in the waste body occurs under anaerobic conditions (Doka, 2003d). This implies that the waste body is consistently covered with adequate covering material and is well compacted. As previously discussed, the biodegradation of organic material in particular is strongly dependent on the conditions of the disposal site, occurring either aerobically or anaerobically under different conditions (Frøiland Jensen & Pippatti, 2001). Accounting for the effect of different landfill operations and management practices on CH_4 generation has been addressed by the IPCC (2006b) with the development of the MCF (see Table 2.9, Chapter 2). Although it is not explicit in the ecoinvent landfill emission model description as to whether the degradation environment that is modelled is fully anaerobic (MCF = 1), if this is assumed to be the case, then the speciation of air emissions determined in the model can be parameterised with the incorporation of the MCF. The proposed modification to incorporate the MCF into the ecoinvent landfill emission model is detailed in Appendix A (Section A.7).

5.3.2 Climate Related Factors

Different climatic conditions such as temperature and precipitation not only affect the degradability of waste, but also affect how these emissions are released from the waste body. The extent to which the existing model can be modified to take into account the effect of climate on the generation and release of landfill emissions is discussed in the following sub-sections.

5.3.2.1 Waste Degradability

Both LFG and leachate generation depend on the extent of degradation of the waste body. According to Doka (2016), several models exist to predict annual CH_4 emissions arising from landfill sites with important parameters in this regard being the speed of decay and the ultimate convertible amount of decay (L0), both of which reflect a climatic dependence. However, it has been suggested that the climatic influence on the speed of decay is not an important factor within LCA modelling (Doka, 2016). This assertion is based on the current resolution of landfill emissions into short- and long-term emissions. Even for very slow decay speeds, the CH_4 generation phase will be unlikely to exceed 100 years and thus will still fall within the short-term time frame defined for the landfill emission model (Doka, 2016). Correcting for the speed of decay will therefore have no discernible effect on the model output.

The effect of climate on L0 by contrast is of relevance to LCA modelling. However, the effect thereof is not limited to methanogenic decay. Within the ecoinvent sanitary landfill emission model, Doka (2003a) uses L0 values to estimate the overall degradability of different waste materials (Doka, 2016). Waste degradability is an important parameter in determining emissions from the waste body, as shown in Equation 5.2 – Equation 5.5 (Section 5.2.4). Hence, the climatic dependence of L0 introduces a climatic dependence into the model output. According to Doka (2016), both humidity and temperature are important parameters affecting L0. Dry climates can significantly reduce L0, while high temperatures favour degradation, and sufficiently low temperatures reportedly prohibit methanogenic conversion altogether (Doka, 2016).

Although the effect of climate on waste degradability is not currently parametrised within the ecoinvent sanitary landfill emission model, investigation into the parametrisation potential thereof has been undertaken by Doka (2016). The proposed parameterisation of waste degradability resulting from this investigation is summarised in Appendix A (Section A.8). While the proposed adaptations show that incorporating the effects of climate into the ecoinvent emission model is possible, implementation thereof requires a detailed understanding of the model and country specific climatic data. Therefore, from the perspective of adaptations that can be undertaken by a product designer or LCA practitioner within the context of a product LCA, the proposed modifications are too complex and time-consuming to be considered feasible. However, with sufficient country-specific data, the existing model could be modified to allow a user-specified waste degradability. For example, a country or region could be a required user input in the model, with this specification linked to predetermined degradation factors. Such a modification would support the generation of a country-specific inventory. The results presented in Doka (2016) suggest that such a model has been developed, however it is uncertain which geographies are represented. Correspondence with the author confirmed that such parametrisation has been undertaken, however the model is not yet publicly available. It is anticipated that this functionality could be seen in future releases of the ecoinvent landfill model.

5.3.2.2 Leachate Generation and Release

In terms of climatic influence, leachate volume is primarily affected by precipitation and the rainwater infiltration rate into the landfill site. Within the ecoinvent sanitary landfill emission model, the mean annual leachate output, V, is assumed to be constant and determined from generic landfill data for landfill height (20 m), waste density (1000 kg/m^3), and rainwater infiltration rate (0.5 m^3/m^2 year) (Doka, 2003d:55). Although changing these parameters within the model will have an effect on the available leachate volume per kilogram waste, this will not affect the emitted pollutants (kg/kg waste), which are dependent on the waste-specific degradability and release factor. The effect of modifying the leachate volume within the model will therefore not incur any additional waste-specific impacts, but has the potential to modify the base burdens used to represent the WWTP process (see Section 5.2.4.2).

For leachate emissions, in addition to waste degradability, the effect of climate can be seen in the release factor. According to Doka (2003d), release rates are dependent on conditions within the landfill body and are strongly influenced by climatic factors, in particular precipitation. The release factors used in the ecoinvent emission model were calculated from "highly variable field data", incorporating a range of landfill sites with different waste compositions, climate, and infrastructure etc. (Doka, 2003d:47). Even with this variability, the results obtained from these calculations were judged as making "*chemical* sense" for the majority of elements (Doka, 2003d:47). It is thus unlikely whether improved estimates for South African specific release factors could be attained without detailed study.

An additional effect of climate is observed in the fate of landfill leachate, which according to Doka (2016) exhibits distinct differences under different climatic conditions. In wet climates, pollutants contained in the leachate are transported downwards towards groundwater ("positive infiltration") (Doka, 2016:9). In dry climates, by contrast, strong evaporation at the surface creates an upward pressure gradient, drawing leachate to the surface ("negative infiltration") where it evaporates, leaving behind precipitates from the dissolved elements (Doka, 2016:9). The resulting precipitates are referred to as "evaporites" and are prone to surface erosion by wind (Doka, 2016:9).

Within the current model, the effect of climate on the fate of leachate is not parametrised. However, a modelling approach to represent this effect has been proposed by Doka (2016). This approach is based on using the net annual infiltration (mean annual precipitation minus actual annual evapotranspiration) to determine whether a landfill site has reversed leachate flow. For sites with a negative net infiltration, it is assumed that no collection and treatment of leachate can be undertaken and evaporites accumulate on top of the site, from where it is assumed that 10% are eroded by wind (inventoried as air emissions) with the remaining 90% inventoried as an emission to industrial soil (Doka, 2016:10-11). Similarly to the case of climatic influence on waste degradability (Section 5.3.2.1), the results presented in Doka (2016) and subsequent correspondence with the author suggests that the ecoinvent landfill emission model has been modified to parametrise the effect of climate on leachate release. Although not currently publicly available, it can be surmised that this functionality is something that could be seen in future releases of the ecoinvent landfill emission model.

Although large areas of South Africa are arid, semi-deserts, the majority of the population is distributed through more temperate areas, suggesting that the majority of waste will be generated in areas with net positive infiltration. Furthermore, given that the effect of net negative infiltration is particularly noted for inorganic landfills (Doka, 2003d), the necessity of incorporating this parameter into the existing ecoinvent sanitary landfill model is debatable. The necessity of accounting for negative net infiltration in South Africa requires further investigation, such as an analysis of the distribution of landfill sites, and determination of actual annual evapotranspiration and mean annual precipitation occurring at each site.

5.4 Other Factors Affecting Landfill Impacts

Although it is recognised within the documentation for the ecoinvent landfill inventory that there is a global disparity in the nature of waste disposal practices, with informal waste management practices being "commonplace" in certain countries or regions (Doka, 2003c:15), such practices are not reflected in the resulting inventories developed for waste disposal. Given that the ecoinvent v3.3 sanitary landfill dataset represents well-managed sanitary landfill practices, the framework for the underlying model has limited capacity for the representation of informal or mismanaged landfill practices.

While the influence of poor management on the emissions from the waste body can be approximated to a certain extent with the modification of the underlying emission model, this approach is by no means exhaustive in representing the impacts associated with informal or mismanaged practices. The following section provides an overview of additional parameters that are not considered in the current ecoinvent modelling framework or landfill datasets, but are of potential relevance to determining the potential impacts of non-engineered landfill sites and open dumps.

5.4.1 Landfill Fires

Landfill sites present a fire hazard due to the high quantity of combustible material on site. Potential ignition sources include conventional sources (i.e. cigarettes or hot ashes), in addition to the high temperatures reached in the waste body during microbial decomposition (Doka, 2003d). Landfill fires can occur either on the surface or underground. While fires at the surface are typically easily detectable, underground fires are harder to detect and can continue for extended periods (Doka, 2003d). It has been suggested that spontaneous underground fires are indicative of poor landfill construction and management, such as insufficient compaction, which results in the formation of flammable methane pockets within the waste body (Doka, 2003d).

The burdens associated with the uncontrolled incineration of waste can be significant (Doka, 2003d). Uncontrolled incineration has the potential to release a number of air pollutants, including nitrogen oxides, CO, VOCs and dioxins in addition to posing an explosion risk due to the presence of CH_4 rich LFG (Doka, 2003d). Due to the significant burdens associated with landfill fires, it has been suggested that including these burdens in the emission output from a landfill site could be relevant despite the rarity of their occurrence (Doka, 2003d). These burdens are not currently included within the ecoinvent sanitary landfill inventory due to the dearth of accurate frequency data on the occurrence of fires on Swiss landfills (Doka, 2003d).

For informal and mismanaged landfill operations, intentional open burning of waste can be common practice (Doka, 2003c). Where open burning is common practice, accounting for these burdens in the landfill inventory becomes relevant. Although the determination of these burdens is not possible within the existing framework for the sanitary landfill model, it has been suggested that waste-specific emissions from landfill fires could be estimated using an inventory model for uncontrolled incineration (Doka, 2003d). Incorporating the resulting inventory effectively into the landfill model would require reliable data with regards to the extent and frequency of landfill fires. Given that such data is lacking for well-managed sanitary sites, this is unlikely to be available for informal or mismanaged sites.

An additional consequence associated with uncontrolled fires on sites with inadequate access control is the risk it poses to waste scavengers. Landfill fires can result in the formation of cavities in the compacted waste body, rendering the site unstable and hence posing a serious risk to scavengers working on the site (Doka, 2003c). Accounting for such impacts poses a challenge to conventional environmental LCA, as indicators for impact categories such as human health do not include occupational hazards. These impacts may thus be better suited for inclusion in a social LCA.

5.4.2 Waste Scavengers

Waste scavenging occurs with the objective of recovering items of economic value from solid waste. Scavenging is typically illegal in countries where “efficient and effective urban solid waste management exists” (Afon, 2012:664). Waste scavenging predominantly occurs in developing countries with poor waste management practices, providing a means of survival for disadvantaged populations (Medina, 2000). Despite the potential economic benefit scavenging provides, scavengers are frequently exposed to a multitude of hazards through their daily contact with waste (Medina, 2000). There are various risks associated with waste scavenging, including occupational risks, physical risks, chemical risks, ergonomic risks, psychological risks, and biological risks (Van Eerd, 1997).

While the activity of scavenging poses a number of risks to the individual, there are certain benefits to this activity that should be considered. The potential income generation is one such example, improving both the economy of the individual and the neighbourhood (Afon, 2012). Furthermore, scavenging plays an important role in reducing the quantity of waste that is landfilled and functions as a supply chain for the provision of raw materials to recycling industries (Afon, 2012). Capturing the impacts of waste scavengers poses a challenge to conventional environmental LCA, as indicators for impact categories do not encompass such risks as occupational, ergonomic, and psychological risks. Given the impact of the waste scavenger on the broader community, the evaluation of the impacts associated with scavenging is potentially better suited for a social LCA.

5.4.3 Vermin and Other Animal Vectors for Disease

Due to its abundance of food, potential for cover, and relatively high temperatures, landfill sites are attractive to a variety of animals including vermin, birds, and insects such as flies and mosquitos (Aderemi & Falade, 2012). The presence of such animals in and around a landfill site is considered hazardous, due to their potential to act as vectors for disease (Rushton, 2003). In addition to spreading disease, it has been suggested that the action of animals, such as birds, that carry waste off-site incur additional health and amenity hazards to residents in surrounding areas (Aderemi & Falade, 2012). For well-managed sanitary landfill sites, various controls such as daily compaction and covering of the waste body are regarded as an important aspect of landfill management, reducing and managing odours from the site, as well as controlling the presence of vermin and other animal species on site. For informal and mismanaged landfill sites that lack such controls, the presence of vermin and other animal species is likely to increase, increasing the risk associated with this hazard (Doka, 2003c).

While adverse effects on human health present one potential impact associated with this hazard, from an impact assessment perspective, considering the full range of potential impacts could include adverse effects on local ecosystems. As in the case of waste scavengers, the presence of vermin and other animals cannot be directly translated into a quantifiable impact for inclusion in a LCI. The inclusion of the impacts associated with uncontrolled animal populations on landfill sites depends on the development of suitable indicators, such that it can be translated into a measurable environmental impact. Due to limitations in being able to represent this burden quantitatively in an environmental LCA, this hazard could be measured qualitatively or else be included as a consideration in a social LCA.

5.4.4 Odour, Dust, and Litter

Where landfill sites lack LFG collection and management systems, LFG is emitted directly into the air. The ecoinvent landfill emission model provides a detailed inventory for these air emissions, from which the environmental impacts can be determined during the LCIA stage of the LCA. While this approach enables the quantification of environmental impacts associated with the various chemical constituents of LFG, it does not take into account additional impacts such as its odour.

LFG contains various compounds, which are associated with pungent odours, of which hydrogen sulphide (H_2S) is typically emitted with the highest rate and concentration (Aderemi & Falade, 2012). Odours continue to emanate from the waste body beyond the closure of the site, and hence can migrate into surrounding communities both during and after the active operation of the site. The malodourous nature of these emissions has the potential to lower the quality of life for residents in these areas (Aderemi & Falade, 2012). An investigation into the air quality and pollution potential of the Robinson Deep Landfill Site in Johannesburg, South Africa, was undertaken by Bhailall (2015) through air dispersion modelling of landfill pollutants. Comparison of measured H_2S concentrations to the World Health Organisation (WHO) guidelines (used due to the lack of a South African national standard for

H_2S) confirmed the landfill site to be a major contributor towards H_2S emissions in the area. However, ambient H_2S concentrations were found to be below the stipulated guidelines (Bhailall, 2015). Based solely on the number of odour complaints received from residents and businesses in areas surrounding the Robinson Deep landfill site, odour modelling was also undertaken, with the results showing the odour impact from the site to be significant (Bhailall, 2015).

The lack of barrier and containment systems present on site further results in the emissions of both dust and litter. Poor compaction and covering, inadequate cover material, and poorly maintained roads can all contribute towards increased dust and litter emissions. Although hazards associated with uncontained dust and litter are less easily quantified than hazards associated with a known chemical element or compound, it is suggested that these might nonetheless impact on communities living in areas surrounding a landfill site (Rushton, 2003).

5.5 Summary of Findings

Existing ecoinvent inventories for the disposal of general waste are limited to sanitary landfill disposal based on characteristic Swiss conditions. Given that South African landfill conditions range from large, well-managed sanitary sites to open dumps, the extent to which local practices can be represented by existing datasets is limited. Based on the structure of the ecoinvent v3.3 sanitary landfill inventory and its corresponding process record within SimaPro, it is possible to undertake a series of generic adaptations to improve the applicability of the dataset to local conditions. Such adaptations include the removal of irrelevant processes, modifying the input value for the burden, or changing the geographic dependence of a linked dataset. Practically, such modifications are limited to the process-specific burdens associated with the landfill process given the paucity of reliable waste-specific emissions data.

Even in terms of process-specific burdens, the availability of representative data for South African landfill sites is lacking. In particular, without directed primary data collection, quantification of process-specific burdens associated with downstream treatment processes — such as leachate treatment and subsequent sludge disposal — are unavailable. Furthermore, with the exception of electricity, South African specific datasets for representing these burdens are lacking. According to Doka (2003d), these limitations are not necessarily prohibitive in the use of the dataset, given the relative importance of the waste-specific emissions on the LCIA result.

The waste-specific emissions inventoried in the sanitary landfill dataset are dependent on the output from an underlying landfill emission model. While the model structure incorporates a parametrisation of certain aspects of landfill operations — such as the composition of the deposited waste and the percentage of gas captured and utilised — the extent to which additional parametrisation is available is limited. These limitations mainly arise over questions of practicality due to the complexity of the data required to undertake such modifications and/or the complexity of the modification. The fact that parametrisation is possible should be considered a benefit of the ecoinvent model. As it is not the intention of the model to be capable of representing alternative landfill geographies or practices, the existing parametrisation is sufficient for its intended purpose which allows adaptation to be made in the context of sanitary landfill operations.

Figure 5.3 overleaf provides a schematic representation of the key findings from this chapter, illustrating the possibilities for current ecoinvent capacity to be modified to represent alternative landfill conditions.

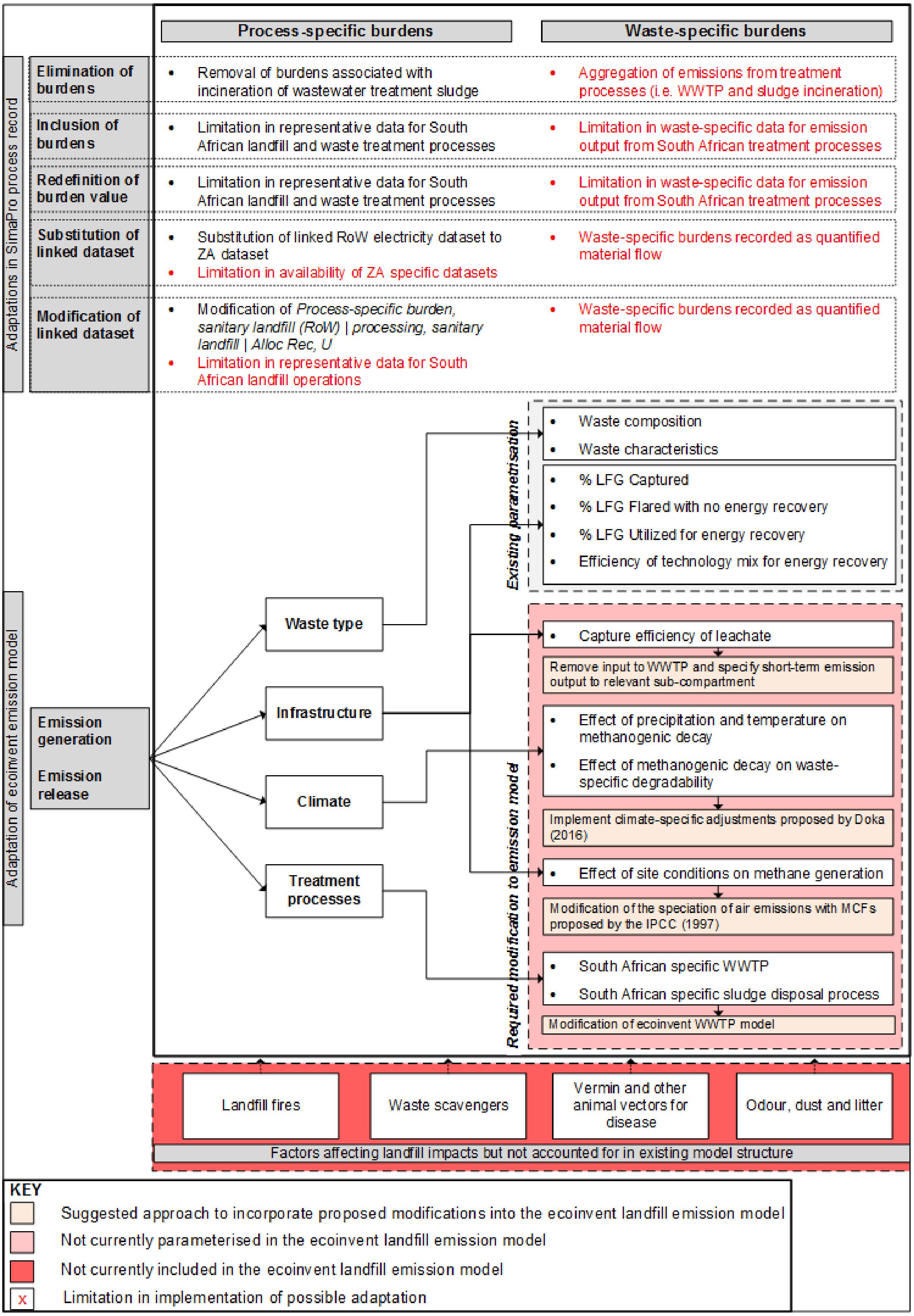

Figure 5.3 Overview of the key findings regarding the adaptability of the ecoinvent v3.3 sanitary landfill inventory to represent alternative landfill practices

Chapter 6

APPROXIMATING END-OF-LIFE IN SOUTH AFRICA BY MEANS OF DATASET MODIFICATION

Modelling product end-of-life in a South African context on the SimaPro-ecoinvent platform is challenging due to the lack of representative datasets for non-sanitary landfill conditions. While various generic adaptations to the dataset are possible due to the modular construction of the ecoinvent v3.3 dataset and its subsequent representation within SimaPro, these are largely limited to process-specific burdens. Waste-specific burdens are generated from the underlying ecoinvent landfill emissions model. Hence, adaptation of the dataset in terms of waste-specific burdens requires modification to the underlying landfill emission model.

The third research question defined for this dissertation is to ascertain the extent to which available landfill models and datasets can be modified to provide a better approximation of landfill disposal in South Africa. While the previous chapter developed an understanding of the current ecoinvent sanitary landfill dataset, emission model, and the parametrisation potential thereof, this chapter explores the effect of these modifications on the LCIA result.

Five different landfill scenarios ranging from well-managed sanitary conditions to open dumps are investigated for three different materials: polyethylene, cardboard, and compostable material. As noted in Chapter 3, the choice of material is intended to represent three non-hazardous materials likely to be found in general waste. Given the temporal dependence of waste degradation, these materials were selected so as to allow comparison of the environmental impact of materials with different organic content and degradability. Due to limitations in the modifications that could practically be undertaken, various approximations and assumptions were necessary. The objective of this chapter is not to provide an accurate quantification of the environmental impacts of different landfill scenarios, but rather aims to compare the different scenarios and identify important parameters in landfill modelling that affect the LCIA result, so as to determine where focus be placed in terms of representing the reality of waste management in South Africa.

6.1 Modifying Landfill Datasets to Represent South African Practises

6.1.1 Definition of Landfill Scenarios

The wide range in South African landfill conditions was represented by the consideration of three different types of landfill sites: sanitary, unsanitary, and open dumps. The ecoinvent v3.3 sanitary landfill dataset for Swiss conditions (CH) (Scenario A) was used as a baseline scenario against which the modified South African specific datasets could be assessed. As discussed in Chapter 5, the system boundary for the ecoinvent sanitary landfill model contains neither the burdens from the waste producing activity nor those associated with the transport of the waste. Therefore, for consistency with the ecoinvent dataset, each landfill scenario utilises the same system boundary. Direct burdens from each process contained within the system boundary include air and water emissions and land use, with indirect burdens arising from energy consumption and infrastructure materials (Doka, 2003d). It should be noted that although the ecoinvent sanitary landfill model assumes that 66% of the captured LFG is utilised for energy or heat recovery, the benefits from this practice (i.e. the avoided emissions from producing electricity/heat by another means) are not included in the SimaPro representation of the dataset. For consistency with this representation, no benefits associated with either LFG recovery or sludge incineration/agricultural application were modelled in the South African scenarios.

An overview of the conditions represented by the five investigated landfill scenarios follows. Based on the results of Chapter 4, together, scenarios B to D represent an estimated 71% of the fate of domestic waste disposed of in South Africa. Scenario E is intended as a proxy for unregulated informal disposal and hence represents an estimated 29% of disposed domestic waste.

<u>Scenario A:</u> Swiss sanitary landfill as represented within the existing ecoinvent sanitary landfill dataset. As detailed in Chapter 5, this scenario represents well-managed Swiss sanitary landfill conditions. According to Doka (2003d), the characteristics of such landfill sites are as follows:

- Base and boundary sealing
- Daily site operations including compaction and covering of waste
- Water collection system
- Gas collection system
- Treatment of collected leachate in WWTP
- Incineration of resulting WWTP sludge in municipal incinerator
- Incineration and/or utilisation of LFG
- Restoration and post-closure monitoring of site

<u>Scenario B:</u> "Best case" South African sanitary landfill. This scenario is intended as a representation of best case sanitary landfill practice in South Africa. It is assumed that the characteristics of landfill sites in this scenario are consistent with those defined for Scenario A with the following exceptions:

- **No utilisation of LFG:** Currently only two landfill sites in South Africa generate electricity (Bhailall, 2015), therefore LFG utilisation was regarded as unrepresentative of landfill practices in the country. Although LFG collection and flaring is also relatively scarce in South Africa (Bhailall, 2015, Friedrich & Trois, 2013b), some landfill sites do have collection and flaring technologies implemented as part of CDM projects, and hence collection and flaring was included to represent best case sanitary landfill conditions in South Africa.
- **WWTP sludge utilised for agricultural application:** Although sludge disposal in South Africa is dominated by on-site disposal methods and landfill (Snyman, 2010), the ecoinvent WWTP parametrisation is limited to representing incineration or agricultural application. As modification of the ecoinvent WWTP model falls beyond the scope of this work, agricultural application was used as a proxy for direct land application. This is discussed in further detail in Section 6.1.2.2.

<u>Scenario C:</u> "Average" South African sanitary landfill. This scenario was intended to represent average sanitary landfill site operations in South Africa. It is assumed that the characteristics of these sites are identical to those represented in Scenario B with the following exception:

- **No LFG capture:** All LFG emissions are released directly to the atmosphere. According to Friedrich & Trois (2013b), the majority of South African landfill sites do not have gas collection systems. Bhailall (2015) reports a total of ten CDM projects in South Africa with LFG collection and flaring/utilisation infrastructure, which are represented by Scenario B. This scenario therefore represents the remaining sanitary sites without collection infrastructure.

<u>Scenario D:</u> Non-engineered South African landfill. This scenario represents average, non-engineered landfill conditions in South Africa. The following characteristics for non-engineered operations are assumed:

- Daily site operations including compaction and covering of waste
- No base sealing
- No water collection system
- No gas collection system

<u>Scenario E:</u> Uncovered, open dump. This scenario represents a “worst case” for landfill disposal and considers unmanaged, open dumps. This scenario is similar to Scenario D, in that it also assumes no LFG and leachate capture, and no treatment infrastructure. However, this scenario further assumes that no daily site operations are undertaken on the site and that all deposited waste is uncovered.

6.1.2 Parametrisation of Landfill Site Conditions

As laid out in the scope for this research, it is intended that the modifications are limited to those that could be considered feasible to be undertaken by a LCA practitioner or product developer within the context of a product LCA. Given the complexity and heterogeneity of landfill systems and a lack of landfill studies for South African conditions, specific waste degradation behaviour was not modified. Although waste degradation is highly variable and influenced by factors including, but not limited to climate, landfill depth, and waste composition (Doka, 2003d), accurate determination of waste degradability and specific landfill system behaviour was considered to be beyond the scope of the study.

Each scenario was developed by means of modification of the existing ecoinvent v3.3 sanitary landfill dataset and emission model. The various adaptations utilised for modelling different landfill scenarios were based on the findings presented in Chapter 5. Process-specific burdens were adapted within the SimaPro process record for the ecoinvent sanitary landfill dataset. Waste-specific emissions, by contrast, utilised the ecoinvent sanitary landfill, incineration, and WWTP emission models. The use of these models generated an emission output that was specified within the relevant SimaPro process record.

6.1.2.1 Process-Specific Burdens

The adaptation of process-specific burdens was undertaken within the SimaPro process record for the sanitary landfill dataset. Given the modular construction of the ecoinvent database, which is supported within SimaPro, the following adaptations were possible:

- Elimination of irrelevant burdens
- Inclusion of additional burdens
- Redefinition of the value of a burden
- Substitution of linked datasets
- Modification of existing linked dataset

An overview of the process-specific burdens modelled for each scenario is available in Table C.4 (Appendix C, Section C.3). Given South Africa’s limited incineration capacity, all process-specific burdens associated with the incineration and subsequent landfill of WWTP sludge in the Swiss landfill dataset (scenario A) were removed from South African specific landfill scenarios, and hence are not shown in Table C.4. The definition of each scenario was in part informed by the results presented in Table 5.2 and Table 5.3 (Chapter 5). Where no South African specific data was available (or the available data considered uncertain) for the quantification of the process-specific demand, the default specified in the ecoinvent sanitary landfill dataset was used. Furthermore, where South African regional datasets were unavailable, the Global (GLO) or Rest of World (RoW) regional datasets were used.

With regards to the modification of existing linked datasets, only modification of the *Process-specific burden, sanitary landfill (CH)* | *market for process-specific burden, sanitary landfill* | *Alloc Rec, U* dataset was undertaken. The changes to this dataset are available in Table C.5 (Appendix C, Section C.3). As before, where no South African specific data was available (or the available data was considered uncertain) for the quantification of burdens, the value specified in the ecoinvent CH dataset was used. Where South African regional datasets were available, the GLO or RoW regional dataset was used to represent the process. It was further assumed that the "Known inputs from nature (resources)" specified in this dataset were applicable for each scenario modelled. These represent the land use burdens associated with the landfill process. As noted in Chapter 5, it can be surmised that the documented burdens are best representative of large, sanitary landfill operations, and hence potentially underestimate unsanitary operations and open dumps. As land use burdens are not a focus of this work, no alternative values were determined.

6.1.2.2 Waste-Specific Burdens

Modification of waste-specific burdens required the modification of the ecoinvent landfill emission model. The ecoinvent emission models are available as downloadable Microsoft Excel files from the ecoinvent database (www.ecoinvent.org) for registered ecoinvent licence holders. The underlying models used to develop South Africa specific inventories for the different landfill scenarios modelled are as follows:

- **Incineration:** 13_MSWIv2.xls
- **Landfill:** 13_MSLFv2.xls
- **Wastewater Treatment:** 13_WWTIv2.xls

The ecoinvent landfill model is linked to the incineration model, with waste specification in only one model to ensure consistency. Hence, the first step in developing inventories for the different landfill scenarios required the specification of waste type and sanitary landfill conditions within the incineration model. Thereafter, changes were made to either the landfill or wastewater treatment models to determine waste-specific emissions under different landfill conditions. The changes made to determine LFG emissions and leachate emissions are discussed in Sections 6.1.2.3 and 6.1.2.4.

A comparison of the key parameters specified for each landfill scenario affecting waste-specific emissions are shown in Table 6.1 overleaf.

Table 6.1 Key parameters considered in the modification of waste-specific emissions for different landfill scenarios

	CH Sanitary Landfill[a]	South African Sanitary Landfill		South African Non-Engineered Landfill	
	Ecoinvent default conditions	LFG Capture and Flaring	Direct LFG Emissions	Covered, unsanitary	Uncovered, open dump
	A	B	C	D	E
Short-term air emissions					
Methane correction factor[b]	Anaerobic, managed MCF = 1	Anaerobic, managed MCF = 1	Anaerobic, managed MCF = 1	Unmanaged–deep MCF = 0.8	Unclassified MCF = 0.6
LFG capture efficiency	47%	75%[c]	0%	0%	0%
LFG emitted directly	53%	25%	100%	100%	100%
Captured landfill gas to flare	34%	100%[d]	n/a	n/a	n/a
Captured landfill gas to utilisation	66%	0%	n/a	n/a	n/a
Total LFG combusted	47%	75%	0%	0%	0%
Short-term leachate (0-100 years)					
Capture efficiency	100%	100%	100%	0%	0%
Leachate treatment	WWTP	WWTP	WWTP	Short-term emissions to groundwater	Short-term emissions to groundwater
Leachate sludge management[e]	Incineration with disposal in slag landfill and residual material landfill	Agricultural application	Agricultural application	n/a	n/a
Long-term leachate (>100 years)					
Leachate emissions[f]	Emissions to groundwater	Emissions to groundwater	Emissions to groundwater	Emissions to groundwater	Emissions to groundwater

[a] Swiss sanitary landfill defaults as reported in Doka (2003d)
[b] MCF assigned as according to the IPCC (2006b)
[c] Due to the lack of local data for time-weighted average LFG collection efficiencies, for consistency with the methodological approach outlined by Friedrich & Trois (2013b), an instantaneous collection efficiency of 75% was assumed. This was based on data from the first South African CDM project in the eThekwini Municipality, which showed that it is achievable in the local context. Therefore, this is presented as a 'best-case' scenario for sanitary landfill operations in South Africa
[d] According to Bhailall (2015) currently only two landfill sites in South Africa generate electricity, therefore LFG utilisation was regarded as unrepresentative of landfill practices in the country
[e] Sludge management for CH dataset based on model default (Doka, 2003d). For South Africa, agricultural application was used given limitations in existing WWTP datasets for representing landfill disposal
[f] Based on the assumption that the failure of base seals and collection systems is inevitable and for long timespans, leachate will enter the ground below the landfill

6.1.2.3 LFG Emissions

As discussed in Chapter 5, the current model capacity is well-suited for modification in terms of LFG emissions. Using the existing ecoinvent landfill model parameterisation, the LFG capture, flaring, and utilisation efficiencies were specified for each scenario (see Appendix C, Section C.2).

Although not parametrised in the current ecoinvent model structure, the effect of different waste degradation environments on carbon speciation was accounted for with the incorporation of the MCF into the model structure. This modification is outlined in Appendix A, Section A.7.

6.1.2.4 Leachate Emissions

As outlined in Chapter 5, the parametrisation potential of the ecoinvent emission model was limited in terms of modifying leachate emissions. This can in part be attributed to the construction of the model, where leachate emissions are dependent on the use of burden factors, derived from the independent WWTP model. It should be noted that no changes were made to the WWTP model with regards to either material inputs or operation. It was assumed that the WWTP itself (excluding sludge management) was comparable with South African municipal wastewater works. Although this assumption requires validation, a detailed investigation into the nature of South African wastewater treatment facilities was beyond the scope of this project.

Therefore, to capture the variation in leachate capture and treatment occurring on South African landfill sites the following considerations were necessary:

- **Leachate collection efficiency:** Leachate collection efficiencies on South African landfills range from 100% on well-managed sanitary sights to 0% on open dumps.
- **Final sludge disposal methods:** Wastewater treatment sludge disposal in South Africa is dominated by direct land application and stockpiling on site.

To account for a reduced leachate collection efficiency, the short-term leachate emission output from the sanitary landfill model was isolated and diverted from the predefined leachate treatment specified in the model. The isolated output then represents the pollutant load with the potential to enter the groundwater beneath the site. Modelling this output poses a challenge due to the temporal dependence of the movement of the pollutant front. According to Doka (2003d), from a long-term perspective, the majority of pollutant is likely to reach the groundwater. For short-term emissions however, the full pollutant load is unlikely to reach the groundwater over the 100 year time frame considered (Hellweg, 2000). For the purposes of this investigation it was assumed that all short-term leachate emissions would reach groundwater within the 100 year time frame. The procedure for determining these emissions is shown below.

1. Short-term leachate emissions were isolated from the ecoinvent sanitary landfill model (Y14 on sheet "MSWLF calculation").
2. These emissions were specified into a SimaPro process record as short-term emissions to water, specifying groundwater as the sub-compartment for their release.

The specification of all short-term emissions as direct emissions to groundwater is an oversimplification of a complex system. Given that common LCIA methods make no distinction between short- and long-term emissions in terms of characterisation factors, specification of all direct leachate emissions as short-term emissions to groundwater will have no effect on the overall LCA results. However, this could overestimate the short-term emission potential of the site.

To modify the leachate treatment process in terms of sludge management, it was necessary to remove the burdens associated with the incineration of treatment sludge and replace these with burdens associated with direct land application. The approach is outlined below.

1. Short-term leachate emissions (kg/kg waste) were isolated from the ecoinvent emission model (Y14 on sheet "MSWLF calculation") and used as an input into the ecoinvent WWTP model (13_WWTv2)[14].
2. The ecoinvent WWTP model was modified such that the distribution of digested sludge to disposal (an existing parameter) was set at 100% to agricultural application (H65 and H66 on sheet "input")[15].
3. The resulting WWTP emissions were exported into the relevant SimaPro process record for the scenario[16].

It must be emphasised that both the approximation used for short-term leachate emissions and sludge management represent a simplified, high-level approach. Therefore, the inventory obtained from this analysis cannot be regarded as precise and must be considered as an approximation of different landfill conditions only.

6.2 Overview of Environmental Impacts from Landfilling

The LCIA results for the landfill of three different materials (polyethylene, cardboard and compostable material) are shown in Figure 6.1 – Figure 6.3 overleaf. These results were obtained using the ReCiPe (H) impact assessment method as available within SimaPro, and are shown as the normalised potential impacts for climate change, human toxicity, freshwater ecotoxicity, and marine ecotoxicity impact categories over a short (0 – 100 years) and long (0 – 60 000 years) time horizon. It should be noted that the long-term time horizon is inclusive of short-term emissions. It should further be noted that all figures use a consistent axis scale, with the exception of Figure 6.1. Owing to the relative magnitude of long-term potential impacts of polyethylene, the y-axis value ranges from 0 – 8 mPe, whereas for short-term emissions, the range is from 0 – 0.8 mPe.

Based on the normalised assessment of impacts using the ReCiPe (H) impact assessment method (using the Europe normalisation/weighting set), the impact categories presented in Figure 6.1 – Figure 6.3 consistently reflected the highest impacts across the three materials investigated. Full results of the LCIA showing all impact categories identified in Chapter 3 are available in Appendix A, Section A.9 (Figure A.1 – Figure A.6), with the quantified potential impacts for each category available in Table A.12 and Table A.13.

[14] Input to the WWTP model is required as kg/m^3 waste-water. Short-term emissions from the landfill model are available as kg/kg waste. Doka (2003b) reports leachate emissions as 0.000025 m^3/kg waste *annum. Over a 100 year period, it was assumed that total leachate generation = 0.0025 m^3/kg waste. Leachate composition for input into the WWTP model was therefore determined as follows: short-term leachate emissions (kg/kg waste//0.0025 m^3).
[15] Only incineration and agricultural application provided as options for sludge disposal. Given that landfilling of sludge is no longer legal in Switzerland, as of 2000, it can no longer be considered as a viable disposal option for sludge and thus was not represented in the WWTP model. It should be noted that only the impacts associated with the leaching of pollutants was modelled and no benefit associated with the use of the sludge as a fertiliser were considered.
[16] The output from the WWTP model is in kg/m^3. Therefore, assuming leachate generation = 0.0025m^3 leachate per kg waste over the 100 year time period, the input into the process record was specified as 1/400 of the WWTP model output.

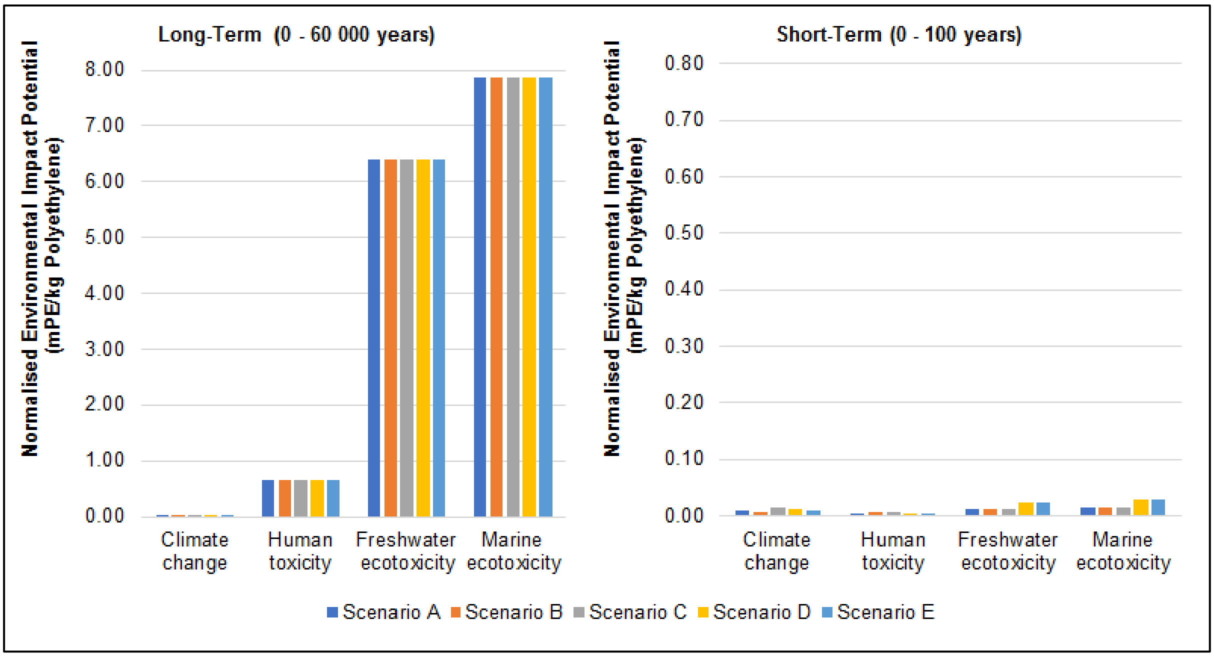

Figure 6.1 LCIA results for the treatment of 1kg of polyethylene in the five landfill scenarios investigated. Results are shown as normalised potential impacts for both a short-term (0 -100 years) and long-term (0 – 60 000 years) time horizon. Note that the long-term potential impacts use a different y-axis range to the short-term potential impacts

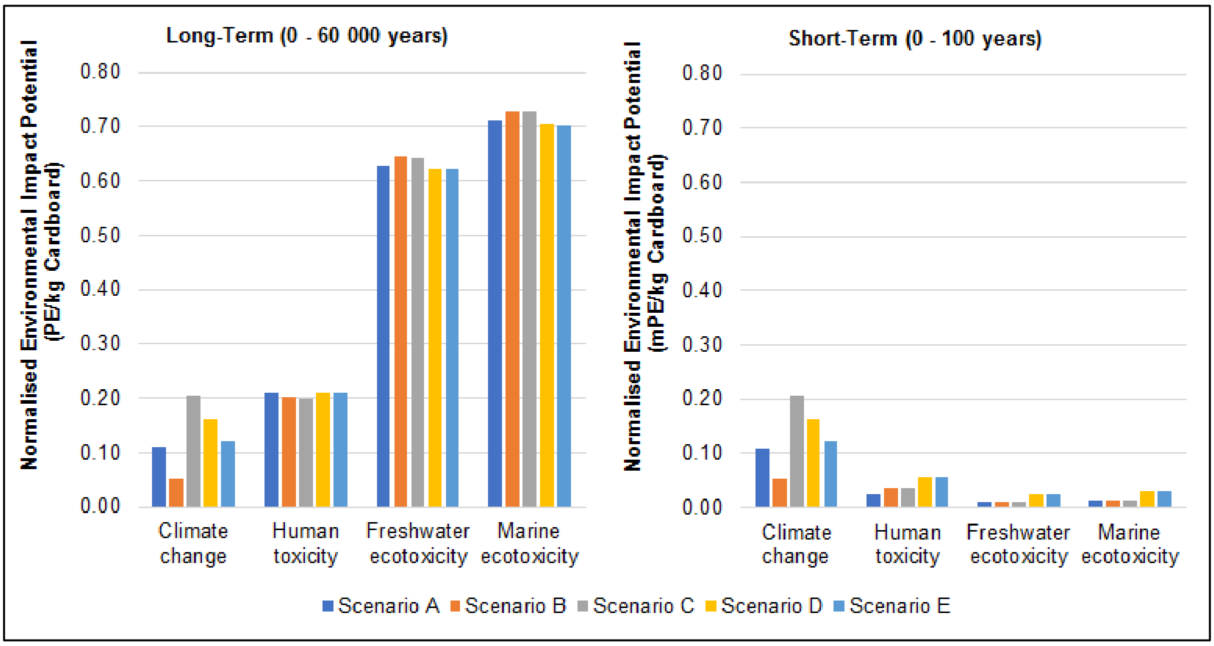

Figure 6.2 LCIA results for the treatment of 1kg of cardboard in the five landfill scenarios investigated. Results are shown as normalised potential impacts for both a short-term (0 -100 years) and long-term ((0 – 60 000 years) time horizon

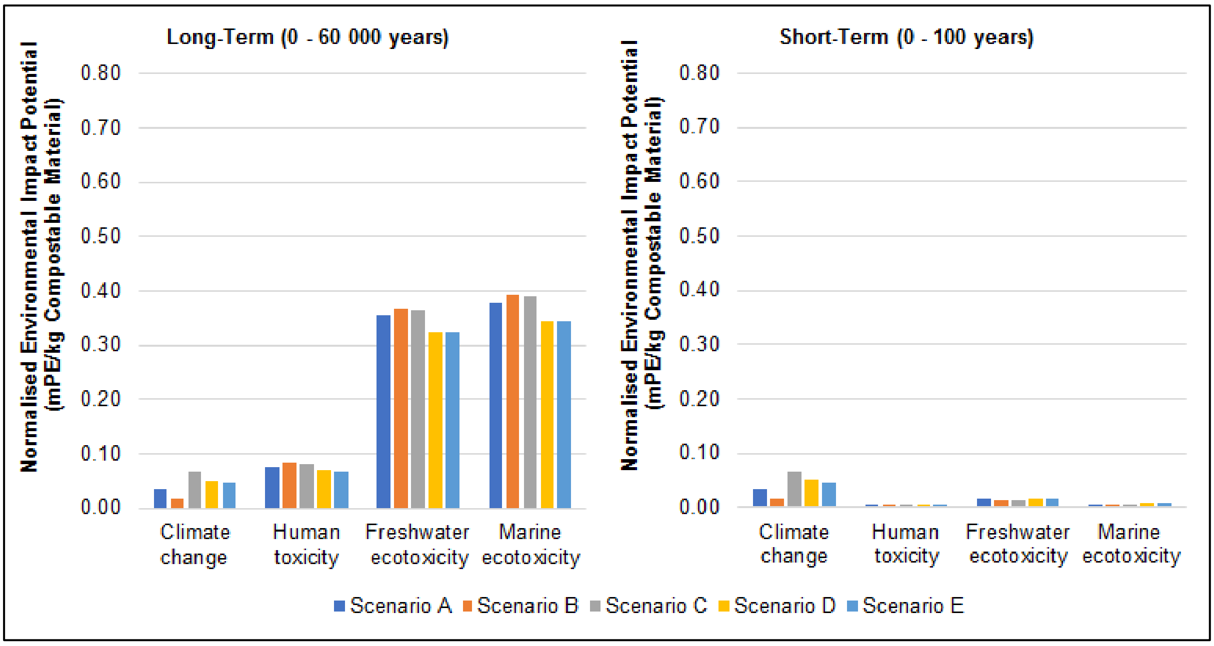

Figure 6.3 LCIA results for the treatment of 1kg of compostable material in the five landfill scenarios investigated. Results are shown as normalised potential impacts for both a short-term (0 -100 years) and long-term ((0 – 60 000 years) time horizon

As illustrated by Figure 6.1 – Figure 6.3, the choice of time frame has a significant effect on the relative environmental impacts associated with landfilling. Regardless of either waste type or landfill scenario, from a long-term perspective, the potential impacts of landfilling is most significant in the freshwater ecotoxicity and marine ecotoxicity impact categories. Evaluation of these categories shows that for all scenarios modelled, the short-term potential impacts contributes less than 5% towards the total potential impacts of these categories (see Appendix A.9, Table A.14). Given that process-specific burdens are largely inventoried as short-term emissions, this result implies that for the waste types considered, waste-specific long-term emissions have the highest potential impacts.

In the ecoinvent landfill emission model, it is assumed that leachate barriers and collection systems fail after 100 years, and hence all subsequent leachate emissions are inventoried as long-term emissions to groundwater. The consequence of this assumption (as illustrated by Figure 6.1 – Figure 6.3), is that regardless of landfill conditions, given the 60 000 year time horizon considered and ensuing extent of waste degradation that occurs during this period, the relative impacts of long-term leachate emissions dominate the environmental potential impacts of landfill sites. Whether long-term landfill emissions are considered or not will therefore have a significant effect on the potential impacts associated with the landfill of a product, regardless of the landfill scenario.

Depending on the material, the relative magnitude of the difference between short- and long-term impacts can vary. Of the three materials considered in Figure 6.1 – Figure 6.3, the landfill of polyethylene shows the greatest variation in potential impacts between the two time frames. The greatest relative increase occurs in the case of freshwater and marine ecotoxicity. While the landfilling of cardboard and compostable organic material also consistently shows a relative increase in potential impacts when considering the long-term time horizon, the relative change is substantially lower than that occurring in the case of polyethylene. Indeed, when assessed on a short-term basis, polyethylene generally shows the best environmental performance. However from a long-term perspective, impacts in these categories far exceed those occurring from the landfill of cardboard or compostable material. This result implies that in terms of product design on the basis of LCA, the choice of material can be more strongly influenced by the time frame considered than the specific landfill scenario.

As discussed in the literature review (Chapter 2), modelling long-term landfill emissions is contentious within the context of LCIA, and addressing the temporal challenges associated with quantifying the potential impacts of emissions remains a topic of interest. While the quantification of the potential impacts of long-term landfill emissions poses a general challenge to representing this process in a LCA, within a South African context, representing the landfill process is further challenged by limitations in obtaining long-term inventory data. Given the climatic variation across South Africa, a relatively high proportion of landfill sites are classified as B^-. This indicates a low leachate generation potential of the site. For the South African specific landfill scenarios modelled in Figure 6.1 – Figure 6.3, long-term emissions are determined using the ecoinvent landfill emission model, and hence represent the long-term emission potential occurring under temperate, Northern European climatic conditions. While this might be accurate for B^+ landfill sites in South Africa (which have a potential for significant leachate generation), for B^- sites the climatic conditions are likely to affect both waste degradation and subsequent emission release. Assuming that leachate emissions from B^- sites are negligible from a long-term perspective, for certain landfill sites in South Africa, all potential emissions could remain *locked-up* in the waste body as opposed to being released as long-term leachate emissions. Removal of the long-term emissions from the landfill inventories developed for each landfill scenario would therefore result in inventories mimicking short-term landfill performance in terms of both emissions and potential impacts. This result implies that landfill disposal in B^- sites has lower potential impacts than thse associated with B^+ sites, where long-term emissions can occur. This effect is investigated in further detail in Section 6.5.

Given that the long-term LCIA results reflect landfill conditions after the failure of leachate collection and containment systems, comparison of the short-time LCIA results arguably provide a better comparison of the performance of different landfill scenarios. Figure 6.4 provides such a comparison, showing the short-term potential impacts in the top four impact categories (climate change, human toxicity, freshwater ecotoxicity, and marine ecotoxicity) of different landfill scenarios for each material.

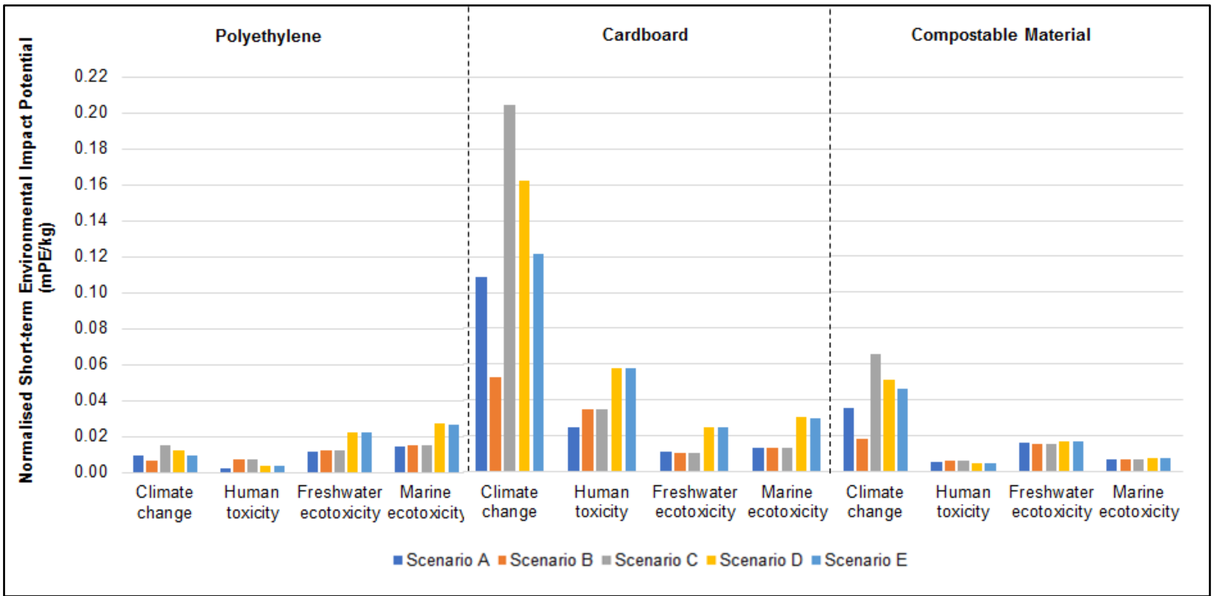

Figure 6.4 Comparison of the short-term (0 – 100 years) potential impacts for the landfill of polyethylene, cardboard and compostable material under different landfill scenarios

Figure 6.4 shows that despite the differences in the relative magnitude of normalised potential impacts between the different materials, in each case, the relative performance of the different landfill scenarios is mostly consistent in each impact category. For both cardboard and compostable organic waste, regardless of the specific landfill scenario, the potential environmental impact of sites is dominated by climate change (albeit to a lesser extent for compostable organic material). For polyethylene, by contrast, the climate change potential is substantially lower, with the most prominent impacts seen in the freshwater and marine ecotoxicity categories. Thus, although the specification of different landfill scenarios has an observable effect on the potential impacts of different categories, the relative magnitude of this effect is strongly dependent on the material type. This suggests that the importance of accounting for different landfill scenarios in a product LCA varies with waste type. However, the end-of-life stage cannot be considered in isolation when undertaking a product LCA and hence, the importance of accounting for different landfill scenarios needs to be considered in conjunction with the other life cycle stages.

6.3 Effect of Different Landfill Scenarios on Climate Change

The results of Figure 6.1 – Figure 6.3 show that in terms of climate change, the relative performance of the different landfill scenarios is consistent regardless of the material considered. Within the ReCiPe Midpoint (H) method, climate change impacts are based on the global warming potential of emissions (kg CO_2 eq). In the context of landfilling, emissions contributing towards climate change predominantly occur over the short-term time horizon. Emissions include LFG emissions from the degradation of waste in addition to various process-specific emissions, such as the burning of diesel on site. Comparison of the relative contribution of waste- and process-specific burdens towards the overall climate change potential of different landfill scenarios shows that for each scenario investigated, the potential impact is dominated by waste-specific emissions. This comparison is shown in Table 6.2.

Table 6.2 Comparison of the relative contribution of short-term (0 – 100 years) waste- and process-specific burdens towards the climate change potential of the different landfill scenarios

	Polyethylene					**Cardboard**					**Compostable Material**				
Scenario	A	B	C	D	E	A	B	C	D	E	A	B	C	D	E
	% Contribution														
Waste-Specific Burdens	92	85	94	97	100	99	95	99	100	100	96	86	96	99	100
Process-Specific Burdens	8	15	6	3	0	1	5	1	0	0	4	14	4	1	0

According to Table 6.2, for Scenario E, for each of the waste types investigated, waste-specific emissions are the sole contributors towards the climate change potential. This result reflects the modelling assumptions used to develop the scenario, specifically that open dumps have no processing requirements. The other scenarios assume certain processing requirements, such as diesel used during the compaction and covering of the waste, and electricity demand for administrative buildings. Given the lack of South African specific data to quantify these demands in the local context, the Swiss default values are used. Although improved local estimates for these demands could theoretically be obtained, given that process-specific burdens typically count for less than 5%[17] of the total potential impact of this category, if landfill impacts are evaluated within the context of a product LCA, this is unlikely to have any discernible impact on the overall results. The results from Table 6.2 therefore suggest that in terms

[17] An exception can be noted for scenario B where the process-specific demands contribute 15%, 5% and 14% for polyethylene, cardboard and compostable material respectively. The relatively high process-specific emissions occur because of low waste-specific emissions due to the 75% capture efficiency assumed for this scenario. However, this scenario reflects a 'best-case' LFG collection scenario which cannot be considered representative of actual local landfill practices.

of climate change related impacts, accounting for the material-specific emissions is a priority in terms of representing different landfill scenarios.

A comparison of the climate change potential for each of the landfill scenarios investigated is shown in Figure 6.5.

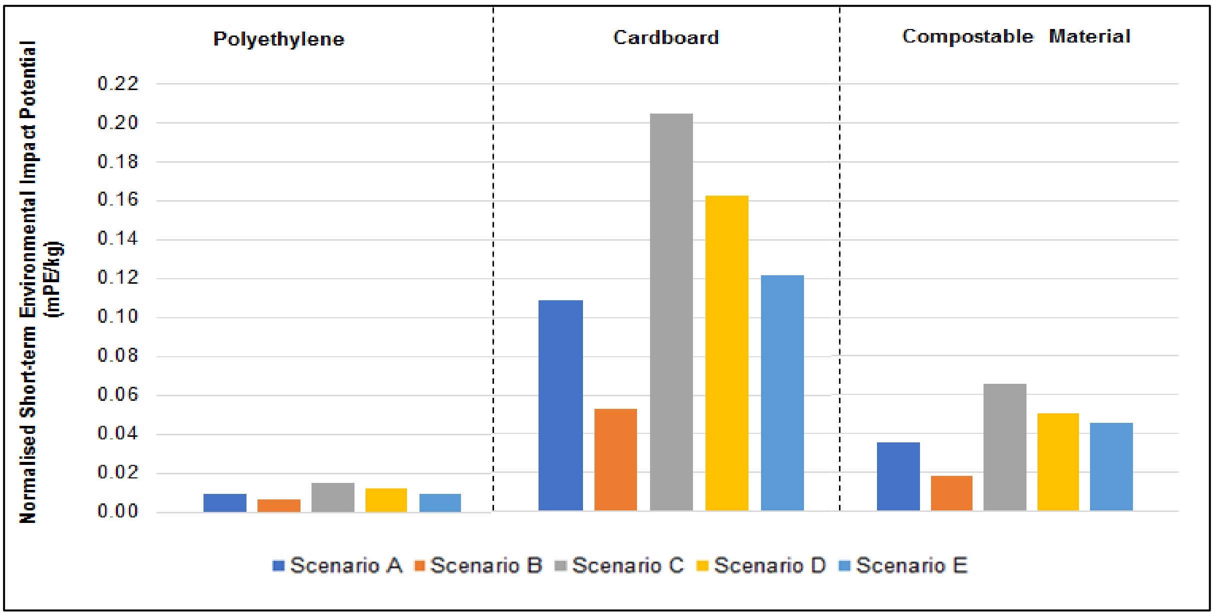

Figure 6.5 LCIA results for short-term (0 – 100 years) normalised climate change potential for the five landfill scenarios investigated

From Figure 6.5 it is observed that both the choice of material and landfill scenario have a notable effect on the climate change potential of landfilling. With regards to the former, LFG emissions occur as a consequence of the degradation of waste, and hence materials with a high degradability and organic component will produce more LFG emissions. According to the material-specific properties used (Table C.6, Appendix C), the degradability of polyethylene, cardboard, and compostable material is defined as 1%, 32.4%, and 27%, respectively. Raw LFG typically contains CH_4, CO_2, CO, O_2, N_2, and H_2S, with CH_4 and CO_2 dominating in terms of composition (Bhailall, 2015, Doka, 2003d). Both CH_4 and CO_2 have a high global warming potential, and hence increased LFG emissions from a waste will increase the potential impact of this category. With regards to the latter, the landfill scenario can also influence the climate change related impacts by means of capture and treatment of LFG emissions and the site-specific conditions capable of influencing waste degradation.

Figure 6.5 shows that Scenario B has the lowest climate change potential for each material considered. This result is somewhat intuitive, as Scenario B is intended as a best-case representation of sanitary landfill in South Africa and models a LFG capture efficiency of 75%, of which it is assumed that 100% is flared. This stands in contrast to the sanitary conditions modelled in Scenario A and Scenario C, representing capture efficiencies of 47% and 0% respectively. The objective of LFG capture and flaring is to combust the CH_4 content of LFG to CO_2, which has a lower global warming potential. As discussed in Chapter 5, LFG capture, flaring and utilisation efficiencies are parametrised within the ecoinvent sanitary landfill emission model, thus allowing the LFG emissions from a range of sanitary landfill conditions to be inventoried with relative ease.

In addition to LFG capture and flaring, the waste degradation environment is an additional parameter capable of influencing the potential impacts of different landfill scenarios. Under anaerobic conditions, waste decomposes to produce CH_4, whereas under anaerobic conditions, CO_2 is produced. According to the *2006 IPCC Guidelines for National Greenhouse Gas Inventories* (IPCC, 2006b), the effect of the waste degradation environment can be accounted for by consideration of a MCF. As outlined in Section 6.1, the ecoinvent sanitary landfill emission model was modified to incorporate a MCF into the carbon speciation (see Appendix A, Section A.7 for further detail). The effect of this modification is observed in Figure 6.5 by comparison of the results obtained for Scenarios C, D and E. For all three scenarios, it is assumed that no LFG collection and flaring is undertaken, and all LFG emissions are emitted directly to the atmosphere. As outlined in Table 6.1, Scenarios C, D and E assumed MCFs of 1, 0.8 and 0.6 respectively. Consequently, Scenario C incurs higher impacts than either Scenarios D and E. This can be attributed to the higher CH_4 content of the LFG as a result of the anaerobic degradation environment assumed in this scenario.

For each of the materials considered, the incorporation of a MCF has a noticeable effect on the climate change potential of different landfill scenarios. Given that the current ecoinvent sanitary landfill model assumes an anaerobic degradation environment, use of this model to represent South African landfill sites overestimates the climate change potential of landfilling. In South Africa, a number of landfill sites are non-engineered landfills or open dumps (see Chapter 4, Sections 4.6 – 4.7), and hence disposed waste is likely to decompose semi-aerobically, resulting in lower CH_4 emissions than would be obtained using the anaerobic ecoinvent landfill emission model. However, although not currently parametrised in the ecoinvent landfill emission model, the current model structure allows the MCF to be incorporated with relative ease into the carbon speciation of LFG emissions.

6.4 Effect of Different Landfill Scenarios on Toxicity Categories

In addition to climate change, the most significant short-term potential impacts of the various landfill scenarios are seen in the human toxicity, freshwater ecotoxicity, and marine ecotoxicity categories (Figure 6.1 – Figure 6.3). A similar analysis to that presented in Table 6.2 is undertaken for each of these toxicity related impact categories to determine the relative contribution of waste-specific burdens towards the overall short-term potential impacts of the category. These results are shown in Table 6.3.

Table 6.3 Relative contribution of short-term (0 – 100 years) waste-specific burdens towards the potential impacts of different landfill scenarios in the human toxicity, freshwater ecotoxicity and marine ecotoxicity categories

	Polyethylene					Cardboard					Compostable Material				
Scenario	**A**	**B**	**C**	**D**	**E**	**A**	**B**	**C**	**D**	**E**	**A**	**B**	**C**	**D**	**E**
	% Contribution of Waste-Specific Burdens														
Human toxicity	61	87	87	96	100	96	95	98	100	100	69	70	70	97	100
Freshwater Ecotoxicity	95	94	94	99	100	94	93	93	99	100	93	95	95	99	100
Marine Ecotoxicity	93	92	92	99	100	92	87	88	99	100	74	73	74	97	100

The results of Table 6.3 show that the contribution of waste-specific burdens is dominant in human toxicity, freshwater ecotoxicity, and marine ecotoxicity impact categories for each of the scenarios considered. The lowest contribution of waste-specific burdens is seen in the human toxicity potential of Scenario A for polyethylene and compostable organic material. Given that process-specific emissions are largely independent of waste type, this implies that the quantified potential impacts from process-specific burdens should be constant for a landfill scenario, regardless of waste type. Therefore, the difference in the contribution of waste-specific burdens towards the overall potential impacts that occurs between different waste types, can be attributed to the properties of the waste itself.

Within the ReCiPe Midpoint (H) method, human toxicity, freshwater ecotoxicity, and marine ecotoxicity impacts are determined as 1,4 dichlorobenzene equivalents (kg 1,4,-DB eq). For Scenarios D and E, short-term waste-specific burdens in these categories are a result of direct leachate emissions. For Scenarios A, B and C, short-term waste-specific emissions are incurred from the treatment of leachate and subsequent disposal of the WWTP sludge. As laid out in Table 6.1, various approximations and assumptions were made using the ecoinvent landfill model in order to develop inventories for landfills with alternative leachate management scenarios. For South African sanitary conditions (Scenario B and C), it is assumed that the WWTP itself is comparable to Swiss conditions (Scenario A), with the exception of the resulting WWTP sludge management. For WWTP sludge management, agricultural application is used as a proxy for landfill in the South African sanitary landfill scenario. Therefore, if only the waste-specific burdens from short-term leachate management are considered, the relative difference in the potential impacts between different sanitary landfill scenarios can be attributed to the differences in sludge management. This is illustrated in Figure 6.6, which shows a comparison of the relative toxicity related impacts for Scenario A and Scenario B/C[18] for the waste-specific emissions arising from the short-term management of leachate.

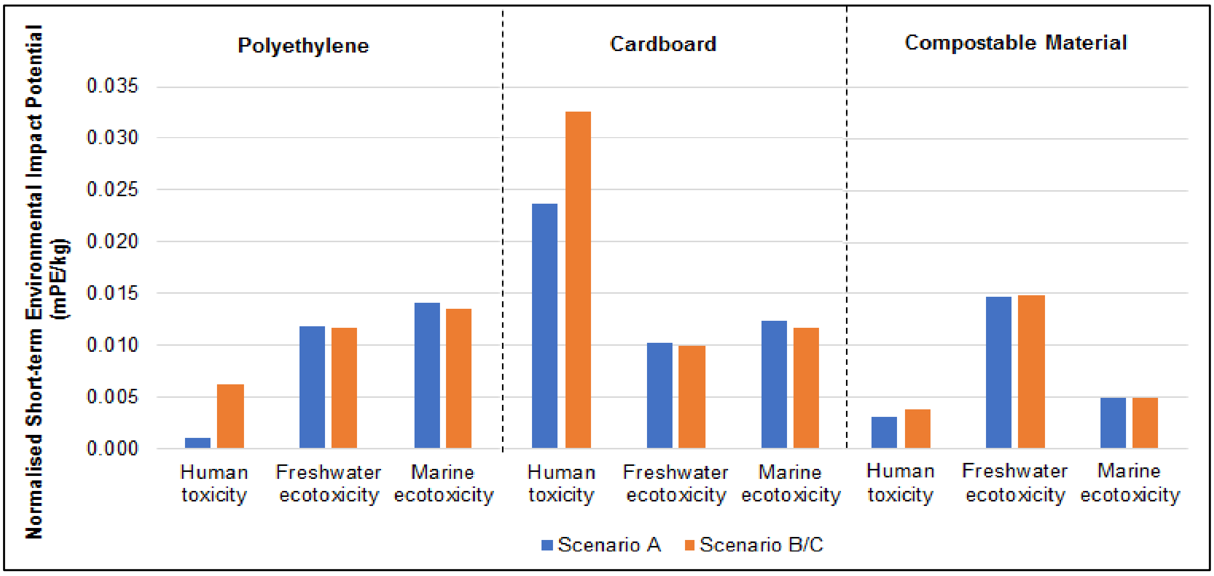

Figure 6.6 LCIA results for the waste-specific burdens associated with the short-term (0 – 100 years) management of leachate shown as normalised potential impacts in the human toxicity, freshwater ecotoxicity and marine ecotoxicity impact categories

Figure 6.6 shows that the most significant difference in the relative impacts incurred as a result of different sludge management occurs in the human toxicity category. For all of the materials considered, the incineration of sludge modelled in Scenario A shows a preferable result in terms of human toxicity potential to the agricultural application of sludge, represented by Scenario B/C. This result suggests that in terms of human toxicity impact, it is necessary to consider the sludge disposal option associated with the leachate treatment from sanitary landfill operations. However, within the context of the total short-term potential impacts of the landfill process, where climate change impacts are dominant, the importance of accounting for this discrepancy is debatable.

For freshwater ecotoxicity and marine ecotoxicity, however, as shown in Figure 6.6, the difference between scenarios is much less pronounced, suggesting that the disposal of sludge has limited influence on the potential impacts of these categories. Furthermore, Figure 6.4 shows that the relative

[18] Scenario B and C were identical in terms of leachate emissions and treatment and hence are represented as a single scenario.

difference in the potential impacts of Scenario A and Scenario B/C in these categories is higher than that illustrated in Figure 6.6. This result suggests that changes to the process specific burdens (for example, changing the geographic representation of the linked dataset) has a greater effect on the overall potential impacts of these categories than changing the sludge management process. This result has positive implications in terms of representing sanitary landfill in South Africa, as the modular construction of the ecoinvent dataset and its representation within SimaPro supports the modification of process-linked burdens preferentially to waste-specific burdens.

For non-engineered landfills and open dumps, as detailed in Section 6.1, it is assumed that all short-term leachate emissions are uncollected and emitted directly to the groundwater beneath the site. Given that the potential impact of the leachate is not reduced in any way before release (i.e. not treated in a WWTP or similar), it is intuitive that the short-term potential impacts of Scenario D and Scenario E should be higher than that in Scenario A, B and C for categories such as human toxicity, freshwater ecotoxicity, and marine ecotoxicity. This result can be observed in Figure 6.1 – Figure 6.3.

While approximating all short-term leachate emissions as direct emissions to groundwater provides an indication of the potential impacts of non-engineered landfills and open dumps, this approach can only be considered a high-level approximation. Accurate determination of the potential impacts of leachate emissions from such sites faces various constraints. For example, modelling the movement of leachate through the soil beneath the landfill site is challenging; it is an oversimplification of a complex system to assume that all emissions will reach groundwater within the 100 year time frame specified. Furthermore, there are various sources of uncertainty in determining toxicity potentials.

6.5 The Effect of Climate on Waste Degradation and Related Impacts

In addition to the physical infrastructure of a landfill site, climate has been highlighted as an important parameter with the potential to affect waste degradation. Based on the discussion above, for both climate change and various toxicity related impact categories, waste-specific emissions dominate the overall potential impacts of each landfill scenario. This implies that under different climates, waste degradation will have an important effect on the overall impacts of the site. As discussed in Chapter 5, although various modifications have been proposed by Doka (2016) to incorporate the effect of climate on landfill degradation, these require both detailed collection of climate specific data and an extensive understanding of the underlying landfill model. Given that this is not a task likely to be undertaken by a LCA practitioner or product developer within the context of a product LCA, incorporation of these factors was not undertaken for any of the scenarios modelled.

However, accounting for various climatic effects — such as precipitation — is potentially relevant in representing South African landfill conditions. In comparison to the temperate Northern European climatic conditions represented in the ecoinvent landfill emission model, the South African climate is on average both hotter and drier (see Figure A.10, Appendix A.9). Without sufficient precipitation, waste degradation is reduced, and emissions cannot be emitted from the waste body. Hence, the potential impacts of the waste remains locked-up in the landfill. In South Africa, given the lack of precipitation that occurs over large regions in the country, as discussed in Chapter 4 (Section 4.6), a high proportion of landfill sites are classified as B^-, signifying that only sporadic leachate generation is likely to occur from the site. It is therefore a pertinent question as to whether leachate emissions are significant contributors towards landfill impacts within the South African context.

Assuming that leachate generation is indeed negligible within the South African context, to investigate the effect this would have on the relative potential impacts of landfill operations, Scenarios B – E are modified to remove any emissions from the leachate itself or process-specific burdens incurred from leachate management. The LCIA results from these modified scenarios are then compared to the long-term LCIA results for scenarios including leachate related burdens (the baseline scenarios shown in Figure 6.1 – Figure 6.3). The LCIA results for the modified scenarios are presented in Appendix A.9, Figure A.7 – Figure A.9. The relative change in the potential impacts of landfill sites without leachate generation or treatment to the original (baseline) scenarios for the different materials considered are shown in Figure 6.7 – Figure 6.9.

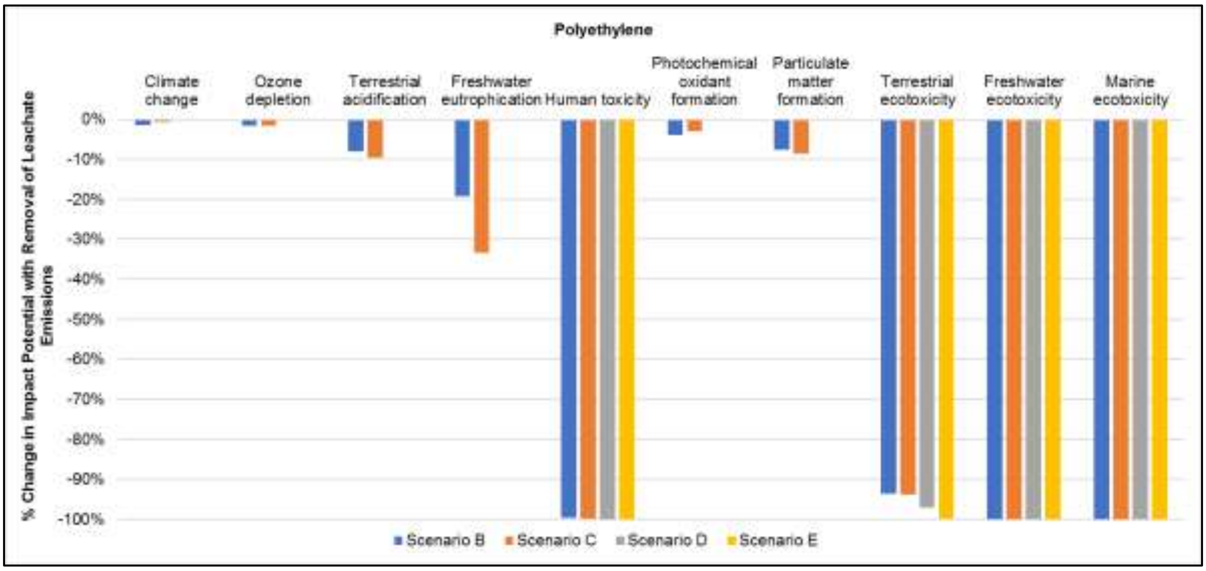

Figure 6.7 Percentage change in the potential impacts of the landfill of polyethylene assuming no leachate generation relative to the baseline scenario

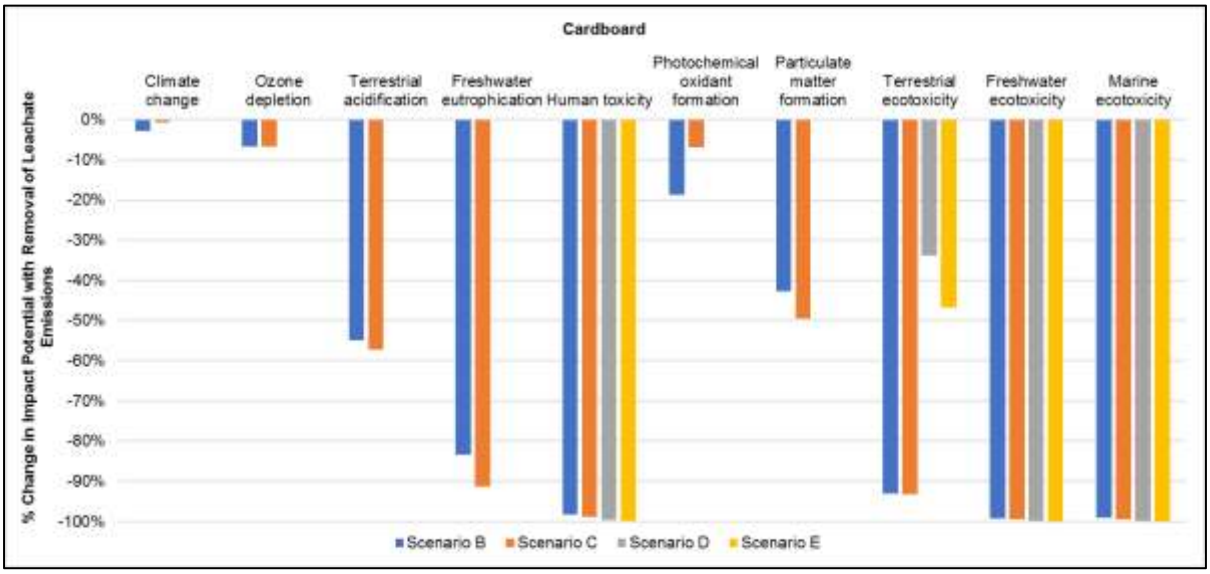

Figure 6.8 Percentage change in the potential impacts of the landfill of cardboard assuming no leachate generation relative to the baseline scenario

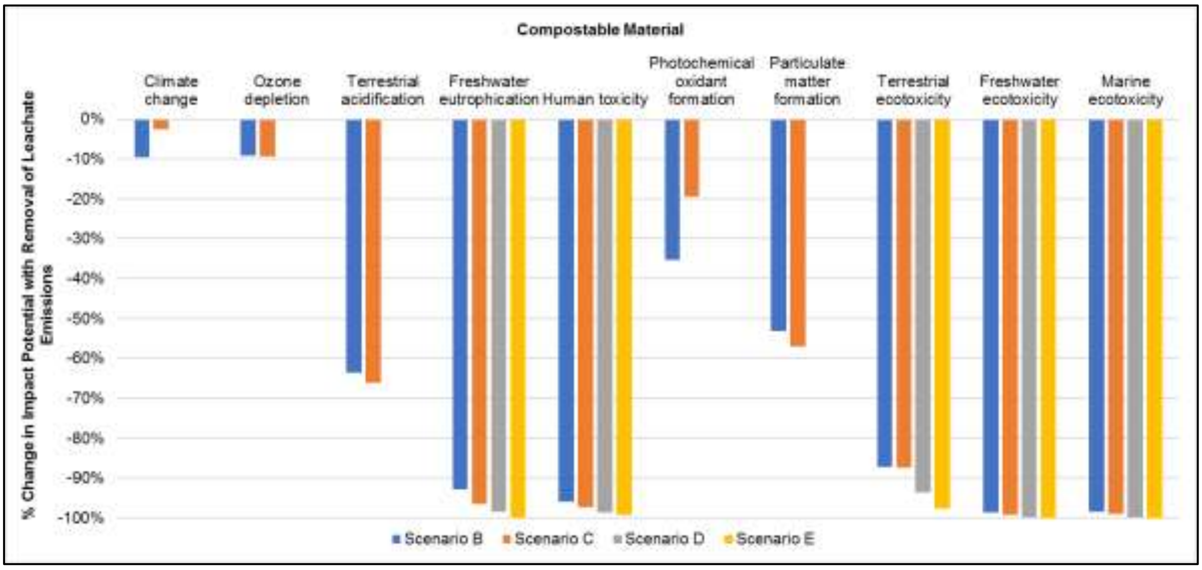

Figure 6.9 Percentage change in the potential impacts of the landfill of compostable material assuming no leachate generation relative to the baseline scenario

The results in Figure 6.7 – Figure 6.9 show that for all three materials, the exclusion of leachate related burdens result in a decrease in the relative potential impacts of all impact categories for the majority of landfill scenarios considered. The exception to this trend occurs in Scenarios D and E, where the removal of leachate related burdens is found to have no effect on a number of impact categories. Given that Scenarios B and C show a relative change in all impact categories, where no changes for Scenarios D and E are observed, it can be assumed that impact reductions in these categories reflect the removal of burdens associated with leachate management. The relative importance of waste-specific emissions towards the overall potential impacts of landfill sites implies that categories dominated by waste-specific emissions are likely to show a more pronounced difference in terms of their quantified potential impacts when modelled with and without leachate related burdens.

Regardless of either landfill scenario or material, the removal of leachate emissions consistently has the highest effect on freshwater ecotoxicity and marine ecotoxicity. Although for all three materials the removal of leachate related burdens has a significant effect on the relative human toxicity and terrestrial ecotoxicity impact, in terms of quantified change, these are substantially lower than those incurred in the freshwater and marine ecotoxicity categories (see Appendix A.9, Figure A.7 – Figure A.9).

Figure 6.7 – Figure 6.9 further shows that the removal of leachate related burdens has only a small effect on the climate change potential of different landfill scenarios[19]. Therefore, compared to other impact categories, the relative magnitude of climate change related impacts will remain relatively consistent, regardless of whether leachate is modelled or not. As illustrated by Figure A.7 – Figure A.9 (Appendix A, Section A.9), with the removal of leachate emissions, climate change becomes the most significant impact associated with all four landfill scenarios investigated. While the choice of material influences the relative magnitude of these impacts, in all cases considered, when leachate emissions are excluded, climate change represents the dominant impact.

[19] The absence of any change in the relative impacts in this category for scenarios D and E suggests that the reduction in potential impacts observed in scenarios B and C reflects the removal of process-specific burdens associated with leachate management.

6.6 Case Study Summary

This chapter investigates the effect that modification of the existing ecoinvent landfill emission model and dataset has on the LCIA result. Five different landfill scenarios ranging from well-managed sanitary conditions to open dumps are investigated for three different materials: polyethylene, cardboard, and compostable material.

Key results from this investigation show that the choice of time frame has the most significant effect on the potential impacts of landfilling, in particular on the freshwater and marine ecotoxicity impact categories. This result is consequent of the assumption utilised in the ecoinvent landfill emission model, namely that leachate barriers and collection systems fail after 100 years, with all subsequent leachate emissions inventoried as long-term emissions to groundwater. Consequently, regardless of the specific landfill scenario, given the 60 000 year time horizon considered and the ensuing extent of waste degradation that occurs during this period, the relative impacts of long-term leachate emissions dominate the potential impacts of landfill sites. This implies that in terms of product design on the basis of LCA, the choice of material can be more strongly influenced by the time frame considered than the specific landfill scenario. Given the implications of the methodological choice of time frame on the LCA result, it is imperative that the LCA practitioner work closely with the product steward and other stakeholders to ensure that the implications of this choice are well understood within the context of the product's life cycle and the relative impacts thereof.

From a short-term perspective, climate change, human toxicity, freshwater ecotoxicity, and marine ecotoxicity consistently reflected the highest potential impacts across the three materials investigated. For these impact categories, the contribution of waste-specific emissions is consistently higher than process-specific burdens. Therefore, both LFG and leachate emissions are important to consider in landfill modelling. In terms of LFG emissions, the current ecoinvent landfill emission model is relatively flexible in representing alternative landfill scenarios. However, the extent to which it can be adapted to represent alternative leachate management scenarios is limited.

In recognition of the potential impact of climatic specificities — such as precipitation — on the LCIA result, Scenarios B – E are modified to remove any emissions from the leachate itself or process-specific burdens incurred from leachate management. Results from this investigation show that the exclusion of leachate emissions lowers the potential impacts of the majority of impact categories, with the most substantial quantified reduction observed in the freshwater and marine ecotoxicity impact categories. The removal of leachate related burdens has only a small effect on the climate change potential of different landfill scenarios, which, after the removal of leachate related burdens, dominate in terms of normalised potential impacts.

Given the flexibility of the current ecoinvent model with regards to representing LFG emissions under different landfill conditions and the various challenges associated with modelling leachate emissions, the dry South African climatic and prevalence of B^- landfill sites (signifying sporadic leachate generation potential) could be advantageous in terms of modelling landfill impacts in a South African context. However, the practical application of this observation is somewhat limited, as the lack of accurate quantified waste data in South Africa prohibits the extent to which landfill disposal can be mapped. Therefore, it remains uncertain with regards to the quantity of waste managed under B^- conditions, and thus to what extent leachate emissions should be considered in a representative South African landfill scenario.

Chapter 7

CONCLUSIONS AND RECOMMENDATIONS

The growing need to redirect human development and consumption patterns towards a sustainable paradigm has seen the promotion of concepts such as life cycle thinking (LCT). Consequently, the life cycle management (LCM) of products has become a regulatory requirement in many countries, informed by the instrument of life cycle assessment (LCA). However, penetration of LCA into developing countries is limited, hampered by such factors as a lack of technical expertise, reliable data and the inability of LCA to engage with the key issues of developing countries (UNEP & SETAC, 2011). These challenges are particularly prevalent with regards to modelling product end-of-life, since waste management in developing countries is typically very different from practices in countries where life cycle methods are well established.

To improve the use of LCA as a tool to support sustainable product design, it is necessary to develop life cycle datasets and methods that accurately reflect the realities of waste management in many countries. The aim of this dissertation was to address this need within the South African context and extract lessons from this for use elsewhere. The objectives for this dissertation were as follows:

1. Identify the current shortcomings of existing LCA datasets in accurately representing the end-of-life stage of general (non-hazardous) waste in a South African context.
2. Propose modifications to existing datasets to better reflect the realities of waste management in a South African context, ultimately to enable better decision-making by product designers and brand owners, for better environmental outcomes in South Africa.

This chapter summarises the key findings arising from this study. This is followed by concluding remarks reflecting on the study as a whole. Finally, recommendations regarding the implications of the findings and opportunities for further research are identified.

7.1 Summary of Key Findings

To meet the objectives defined for this study, research was undertaken in three main stages, namely (1) an investigation into the status quo with regards to waste management in South Africa, (2) a review of current LCA capacity for representing product end-of-life in South Africa, and (3) an investigation into the implications of imposing modifications to existing datasets to represent alternative end-of-life management practices. The key findings from these stages are summarised in Sections 7.1.1 – 7.1.3.

7.1.1 Establishing the Status Quo for General Waste Management in South Africa

Although waste legislation in South Africa is of a high standard — with the promulgation of the *NEM:WA* (2009) supporting reform in terms of waste reporting and the adoption of the principles of the waste hierarchy for waste management — the uptake of and adherence to this legislation is lacking. Due in part to its departure from traditional waste management systems, for most municipalities, implementation of the Act is complex, requiring changes to both systems and infrastructure, which demands a level of capital funding and changes that are beyond municipal capacity. Devolving responsibility for the actual implementation of waste services to local government only works where there is functioning local government, however this is not the case in many South African municipalities.

The disparity in the provision of formal waste management services complicates the determination of a representative waste scenario for South Africa, requiring the consideration of both formal and informal management scenarios. In the formal sector, developing a quantified mapping of waste flows is constrained by limitations in comprehensive and accurate waste data. Although the SAWIS is intended to provide a national repository for waste data, the system is not yet fully representative, with only a proportion of active facilities reporting data and furthermore, doing so with a questionable accuracy. In lieu of a fully functioning and verified national waste repository, the most comprehensive source of quantified waste data is that contained in the *National Waste Baseline Report* (DEA, 2012a). However, not only are these results somewhat outdated, but also of unspecified precision and partly based on questionable assumptions (von Blottnitz, 2016).

A major limitation in both the SAWIS and the *National Waste Baseline Report* (DEA, 2012a) in providing a quantified national waste estimate lies in their omission of informal waste, thus resulting in the potential under-representation of both waste quantities and waste management practices. Using South African specific waste generation rates (accounting for both income distribution and settlement type), it is estimated that South Africa generates approximately 12.7 million tonnes of domestic waste per annum. Using this result and national service delivery statistics, informal household waste generation is estimated to be in the order of 3.67 million tonnes. This implies that 29% of domestic waste generated in South Africa is not collected or treated via formal management options. Of this waste, approximately 3.13 million tonnes (85%) is generated in rural areas. For all settlement types, the most common waste management option for un-serviced households is a private dump (the use of private dumps for the disposal of informal waste is reportedly 94%, 74%, and 71% in rural, urban, and metro areas, respectively). Illegal dumping is the next most common option and ranges from 5% for un-serviced rural households to 27% in metro areas with the balance made up by “other” disposal/treatment options.

While consistent and accurate estimations of actual waste tonnages are limited, there is general consensus that disposal to land remains the most utilised option for the majority of South Africa’s general waste. However, there is a significant range in infrastructure and operating standards between different sites. At one end of the spectrum, private refuse dumps are unlikely to provide any form of engineering control, hence can be considered little more than open dumps, whereas at the other, formal landfill operations can include well-managed sites with a high level of engineering controls and barrier systems in place to manage and contain emissions. Even for formal disposal, a range of landfill conditions exist. Regulated management and controls for these formal sites are primarily informed by the daily waste deposition and the leachate generation potential of the site, becoming increasingly stringent with increasing size class and leachate generation potential. Based on the distribution of licensed waste disposal facilities and the waste deposition range specified for each size class, it is estimated that approximately 54% of (formal) general waste in South Africa is disposed of in large landfills, 31% in medium landfills, 12% in small landfills and the balance (2.8%) in communal sites.

Whilst formal landfill sites have a set of regulated standards, the extent to which sites are compliant with these standards should also be considered. Analysis of available landfill audit reports show that even if the compliance statuses of landfill sites in the Western Cape — a relatively well-run province in terms of waste management — are used as a proxy for national compliance, the proportion of non-complaint waste disposal facilities is appreciable at 73%. While large landfill sites in the province are fully compliant, the proportion of non-compliant sites increases with decreasing size class (with non-compliance of medium, small and communal sites at 50%, 75%, and 82%, respectively). This suggests that when defining landfill operations in South Africa, it may be necessary to consider the performance of non-compliant sites as part of a representative landfill scenario.

The validity of representing waste disposal in South Africa by means of one generic set of landfill conditions is questionable, given the range in operating standards between different sites. The range in landfill infrastructure and operating standards will influence both the emissions and subsequent environmental impact of the site. For the purposes of evaluating the end-of-life impacts from a LCA perspective, it is therefore questionable as to whether these different sites can be represented by a generic landfill dataset or whether further distinctions should be made with regards to the nature and distribution of site types. For example, given the variation in the landfill sites operating in South Africa, the best representation of disposal might be by means of a “market type” dataset with different types of sites taking different proportions of the disposed waste.

7.1.2 Review of Current LCA Capabilities for Representing Product End-of-Life in South Africa

Within SimaPro software (v8.3 was used), landfill disposal of general waste is best represented by the sanitary landfill datasets contained within the ecoinvent v3.3 database. Although various waste types are inventoried, there is limited geographical differentiation between datasets, with all datasets based on the operation of well-managed, sanitary Swiss landfill practices. The available sanitary landfill datasets inventory the infrastructure, resource and process-specific demands for the landfill process in addition to the waste-specific emissions from the site.

It is useful if LCA modelling software preserves the modular construction of the ecoinvent dataset, meaning that various generic modifications to the inventoried data can be made, such as the elimination or addition of burdens, redefinition of the value of a burden or substitution of a linked dataset. Practically, such modifications are limited to the process-specific burdens. The process-specific demands inventoried in the ecoinvent sanitary landfill dataset are based on actual site data obtained from representative Swiss landfill sites. The burdens associated with these demands are introduced into the landfill inventory by linking the demand to a representative ecoinvent dataset. Therefore, in terms of modification potential, with directed primary data collection, developing quantified South African process-specific burdens is feasible. However, the extent to which South African specific datasets are available within ecoinvent to represent these burdens is limited, with only the provision of electricity currently represented by a regional dataset. This limitation is not necessarily prohibitive in the use of the existing ecoinvent dataset, as it is well documented that waste-specific burdens are typically most significant in the LCIA result of a landfill process (i.e. Doka (2003d), Kirkeby et al. (2007), Manfredi & Christensen (2009)). This implies the relative importance of focusing on waste-specific emissions with regards to adaptation potential.

Waste-specific emissions inventoried in the ecoinvent v3.3 sanitary landfill dataset are generated from an underlying landfill emission model. The current model structure allows for the specification of waste composition in addition to the LFG capture and utilisation efficiencies. With regards to the model’s fitness for purpose, the existing parametrisation is sufficient, as it is intended that the model represent a sanitary landfill process and therefore allows a user to specify key variables affecting emissions within this context. Given the lack of datasets representing alternative landfill practices — such as non-engineered landfills and open dumps — the adaptation of the existing model provides a practical alternative to developing a suitable inventory by alternative means (i.e. developing a model or undertaking primary data collection).

However, besides the incorporation of a methane correction factor (MCF) into the speciation of air emissions to account for the effect that various site conditions have on the waste degradation environment, the extent to which the existing model can be adapted to represent alternative landfill conditions is limited. Again, this should not be regarded as a shortcoming in the model itself, as such capabilities are beyond its intended purpose. Although various adaptations have been proposed by

Doka (2016) to incorporate the effect of climatic conditions on waste degradability and emission release, implementation of this approach requires a high level of country-specific data and modelling expertise. These requirements question the practicality of such a modification within the scope defined for this research. However, correspondence with Gabor Doka (the author of the proposed modifications and the ecoinvent landfill model) suggests that the ecoinvent landfill emission model is undergoing modification to include parametrisation to account for various climatic conditions. Although not currently publicly available, it is anticipated that this functionality is something that could be seen in future releases of the ecoinvent database.

While alternative landfill practices can be approximated to a certain extent within the existing modelling framework by means of modifying LFG capture efficiencies, incorporating a MCF, and approximating all short-term leachate emissions as direct emissions to groundwater, this approach is by no means exhaustive in representing the potential impacts of unmanaged landfill sites. For example, the open burning of waste can occur with appreciable frequency on informal or poorly managed formal landfill sites and give rise to a number of air pollutants. Furthermore, the limited controls on unmanaged and informal landfill sites increase the occurrence of waste scavengers on site and lead to increased odour, dust and litter, and the presence of vermin and other animal vectors for disease. The evaluation of these burdens is excluded from the scope of current inventories and LCIA methods.

Although there is a clear limitation in the availability of datasets representing alterative landfill disposal practices, this is by no means the only challenge facing landfill modelling within the context of LCA. Obtaining accurate, waste-specific landfill emission data is notoriously challenging due to the complexity and heterogeneity of landfill systems. One such complexity is the time frame associated with waste degradation and emission release. LCA typically inventories the total mass of emitted substance, thus excluding temporal dependencies in the inventory data. The inclusion of long-term impacts can therefore be contentious. In lieu of consensus on the use of temporal discounting or other means to quantify the potential impacts of future emissions, the ecoinvent dataset makes a rudimentary temporal distinction, inventorying short-term emissions (0 – 100 years) with long-term emissions (100 – 60 000 years) represented in a separate sub-category (Doka, 2003d). The benefit of the separation is seen in the LCIA stage where, although both sets of emissions are assigned the same impact factor, it allows the landfill impacts to be assessed from both a long- and short-term perspective. However, despite this distinction, defining a suitable time frame for landfill modelling remains contentious. For example, given the ensuing extent of degradation that occurs over 60 000 years, the practise of very long-term modelling could equalise landfills that differ strongly in the short-term. Resolution on how to manage such temporal challenges within this context remains to be reached.

7.1.3 Implications of Dataset Modification on Product End-of-Life

The possibility to modify ecoinvent landfill datasets for the disposal of polyethylene, cardboard, and compostable material was tested under various landfill scenarios. Analysis of the LCIA results obtained using the ReCiPe midpoint (H) method showed that regardless of either the deposited material or the specific landfill conditions modelled, the time frame considered has the most pronounced effect on the potential impacts of a landfill site. Within the current sanitary landfill model framework, it is assumed that after 100 years, leachate barrier and containment systems fail, and all subsequent emissions (occurring over the course of 60 000 years) are inventoried as emissions to groundwater. Over 60 000 years, even relatively slowly degrading materials — such as polyethylene — will have undergone some level of degradation. Consequently, regardless of initial conditions, when long-term emissions are considered, freshwater and marine ecotoxicity impacts dominate the overall potential impacts of the site. This result implies that if landfill disposal is modelled over the long-term, the potential impacts of the process has less to do with site-specific conditions than it does to do with the intrinsic properties of the waste itself. Therefore, in terms of product design on the basis of LCA, the choice of material can be more strongly influenced by the time frame considered than the specific landfill scenario.

Given that the long-term LCIA results can equalise landfills that could differ strongly in the short-term, comparison of the short-time LCIA results provide a better comparison of the performance of different landfill scenarios. From a short-term perspective, site-specific conditions can have a discernible effect on the LCIA result. For cardboard and compostable material (materials with a high degradability), the effect of LFG capture and treatment has the greatest effect on the potential impacts of the site. The impact of LFG emissions is predominantly seen in the climate change impact category but also, albeit to a lesser extent, in photochemical oxidant formation and particulate matter formation. In terms of climate change, sanitary landfill sites without LFG capture have the highest potential impacts, exceeding both non-engineered sites and open dumps. This result reflects the effect of the waste degradation environment on the potential impacts of a site, accounted for by incorporation of a MCF into the air speciation calculation in the ecoinvent emission model. The relative importance of climate change in terms of the short-term potential impacts of landfilling and the effect that the MCF has on the modelled results, illustrates the importance of accounting for the waste degradation environment of different landfill scenarios.

The short-term LCIA results for slowly degrading polyethylene showed different results, with freshwater and marine ecotoxicity in addition to climate change showing the most significant potential impacts for all scenarios considered. Comparison of the relative magnitude of the potential impacts for this material to the results obtained for cardboard and compostable organic material shows the overall impacts to be substantially lower in terms of normalised potential impacts. This result suggests that the relative importance of incorporating site-specific parameters varies depending on the type of material disposed.

When assessed from a short-term perspective, for fast degrading materials the impacts incurred from leachate emissions and their subsequent treatment are of lesser importance (as measured by world-normalised impact scores) than those associated with LFG. Even for non-engineered landfill sites and open dumps, where it was assumed that all short-term leachate emissions are emitted directly to groundwater, this modification has lesser effect than those made to modify the LFG emissions. This bodes well for representing alternative landfill scenarios, given the existing parameterisation of the ecoinvent landfill emission model (LFG capture, treatment and utilisation efficiencies) and feasible modification potential (the incorporation of a MCF into the air speciation calculation).

Waste degradability is an important factor for both LFG and leachate emissions and has a strong climatic dependence. Due to its various complexities, accounting for climatic specificities on landfill degradation falls beyond the scope defined for this research. Accounting for various climatic effects — such as precipitation — is, however, potentially relevant in representing South African landfill conditions. The exclusion of leachate emissions and related burdens from the various scenarios investigated lowered the potential impacts of the majority of impact categories, with the most substantial quantified reduction observed in the freshwater and marine ecotoxicity impact categories. This result implies that for dry climates (such as those in the western parts of South Africa), the long-term potential impacts of landfilling could be significantly lower than when compared to landfill under temperate or sub-tropical conditions, with the potential impacts of the waste remaining locked-up in the landfill.

The removal of leachate related burdens has only a small effect on the climate change potential of different landfill scenarios. Consequently, with the removal of leachate emissions, climate change becomes the most significant impact associated with all landfill scenarios investigated. Given the flexibility of the current ecoinvent model with regards to modelling LFG emissions and the various challenges associated with modelling leachate emissions, the dry South African climate and prevalence of B^- landfill sites (signifying sporadic leachate generation potential) appears to be advantageous in terms of modelling landfill impacts in a South African context. Due to the current limitations in the availability of accurate waste data in South Africa, it remains uncertain as to what extent leachate emissions are relevant in the development of a representative South African landfill scenario.

7.2 Significance of Findings

As per the objectives, this dissertation set out to consolidate existing knowledge regarding waste modelling in the context of a local LCA and where necessary, extend this knowledge by means of adaptations and modifications of internationally developed calculation procedures to better reflect the realities of waste management in a South African context.

In South Africa, waste management is dominated by disposal to land (landfill or dumping). Given the notable disparity in the level of waste service provision across South Africa, a large proportion of waste is disposed of under less-than-sanitary conditions. This proportion comprises both informal waste — generated by un-serviced households relying on open-dumps and other means to manage their waste — and formal waste managed under poorly regulated and controlled conditions. Given the potential variation in the impacts arising from different site conditions, application of LCA methodology to product end-of-life within South Africa requires that the distribution of waste between different disposal sites is known. To date, existing waste data does not reflect the contribution of the informal sector neither does it distinguish between the waste accepted by different landfill classes. The mapping of waste flows undertaken within this dissertation therefore provides an important contribution towards the application of LCA within the local context, providing an estimate of the “market share” of different waste management options.

While knowing the distribution of waste is a necessity within the context of evaluating end-of-life impacts of a product, for a useable outcome within a LCA, each management option must be represented by appropriate LCA datasets. Given the geographic imbalance in the uptake and distribution of LCA methodology, existing datasets are best representative of waste management practices undertaken in developed countries. While these are relatively well suited for representing landfill disposal in large, well-managed sanitary landfills in South Africa, they fall short in representing poorly regulated sites and open dumps. The adaptation and modification of the internationally developed calculation procedures to allow better representation of the local context undertaken in this dissertation is therefore an important contribution towards improving LCA capacity within South Africa, with further application to other developing countries.

Improved local knowledge of waste management and better representative LCA datasets will better reflect the realities of product end-of-life in a South African context, ultimately enabling better decision-making by product designers and brand owners, for better environmental outcomes.

7.3 Concluding Remarks

While the status quo results suggest that waste management in South Africa reflects aspects of the waste management practices typical of developing countries — such as open dumping, variable waste collection services, and limited environmental control systems to manage the environmental impacts of waste — it simultaneously reflects management practices well aligned with legislative requirements. It is therefore anticipated that as waste management becomes increasingly aligned with the aspirational vision of the *National Environmental Management: Waste Act, No. 59 of 2008* (2009), the need for South African specific LCA datasets will diminish, as South African waste management practises fall in alignment with good (if not best) global practice.

In the interim, South Africa's ongoing dependence on landfill and the range in landfill operations active across the country suggests that representation of end-of-life management in South Africa requires alternative datasets to those which are currently available. It is acknowledged that as only one software tool and database platform were interrogated in detail, these results are limited in their view of LCA capacity for representing alternative disposal. Given the various complexities associated with obtaining landfill inventory data and the current lack of datasets representing alternative landfill practices — such as non-engineered landfills and open dumps — the adaptation of the existing ecoinvent landfill dataset and underlying emission model provides a practical alternative to developing a suitable inventory by alternative means (i.e. developing a model or undertaking primary data collection).

With regards to the ecoinvent v3.3 sanitary landfill dataset, while certain generic adaptations to the process-specific burdens can be undertaken, accurately accounting for waste-specific burdens through modification of the underlying emission model should be considered a priority in developing a landfill inventory, due to their relative importance in the LCIA result. In terms of LFG emissions, the existing parametrisation of the ecoinvent sanitary landfill model enables the representation of a wide range of landfill conditions through modification of LFG capture, containment, and utilisation efficiencies. Furthermore, the waste degradation environment can be accounted for with the specification of a suitable MCF within the current model framework. With regards to leachate emissions however, parametrisation potential is limited. On-going developments to current ecoinvent capacity suggest that parametrisation to account for alterative climatic conditions and the effect this will have on waste degradability is something likely to be seen in the future.

From the perspective of product design, given quantified findings on South Africa's dependence on both formal and informal disposal, and the variation in landfill conditions across the country, it can be concluded that LCA results for the impacts of products originating from global supply chains, but consumed and disposed of in South Africa, will be inaccurate for the end-of-life stage if modifications to end-of-life modelling are not made. The findings from this dissertation provide the basis for i) a crude estimate of "market shares" of different disposal practises and ii) guidelines for parameterisation of material specific emission factors, in particular for shorter term emissions, focused on LFG and leachate emissions. Over very long time frames, emissions from landfill disposal to some compartments of the environment are almost entirely material specific and not influenced significantly by the nature of the landfill.

7.4 Recommendations

7.4.1 General Recommendations

While the lack of accurate quantified waste data is a known limitation in terms of mapping waste management in South Africa, without adequate information on waste volumes, composition, and management, representation of the end-of-life stage in a product LCA in a South African context will remain imprecise. It is therefore necessary to improve waste reporting and enforce compliance with existing legislation in the local context, because as waste reporting practices improve, the possibility of accurately including end-of-life in LCA modelling also improves.

In terms of current LCA capacity for representing end-of-life in a South African context, within SimaPro v8.3, the representation of alternative landfill disposal practises is limited to well-managed sanitary conditions. To approximate alternative landfill conditions, adaptations to process-specific burdens can be made by means of generic changes to the ecoinvent v3.3 sanitary landfill dataset. With regards to waste-specific emissions, the existing parameterisation in the ecoinvent v3.3 sanitary landfill emission model can be used in conjunction with the modifications proposed in this dissertation. Given the various limitations in this approach, especially with regards to leachate modelling and accounting for various climate specificities on the waste-specific output, it is recommended that the current parameterisation be used as an interim measure until proposed updates to the sanitary landfill model have been implemented in the ecoinvent database.

On the other hand, while the traditional scope for environmental LCA determines potential impacts in terms of quantified emission flows, the wide range of alternative impacts associated with various alternative landfill practices — such as waste scavengers, vermin, and increased incidences of uncontrolled burning and uncontained litter — suggests the need to broaden the traditional scope and consider a wider range of impacts. Due to limitations in being able to represent such burdens quantitatively in an environmental LCA, the effect of these factors could be measured qualitatively or be included as a consideration in a social LCA. Particularly in South Africa and other developing countries where formal and informal waste management practices co-exist — and indeed frequently overlap as in the case of waste scavengers — assessing only the environmental impacts associated with the disposal of a product could be insufficient in accurately assessing its sustainability. In such cases, a more holistic assessment is necessary, considering environmental, societal and economic factors.

7.4.2 Recommendations for Further Work

The scope of this project limited the detailed interrogation of existing LCA capacity to datasets contained within SimaPro. Within SimaPro v8.3, the landfill process is best represented by ecoinvent datasets. Consequently, the combination of one LCA software tool and database platform was interrogated in detail. Given the availability of alternative LCA software tools and database modelling platforms, it is recommended that this analysis be repeated on additional software tools and database platforms to achieve a broader view of current LCA capacity for representing landfill disposal.

The results of this dissertation highlighted the need for landfill datasets representing a range of site conditions by showing that both climate, and site operations and infrastructure have the potential to influence the potential impacts of the site. However, this investigation is limited to considering the end-of-life stage, and thus does not provide an indication of the relative importance of accounting for landfill site specificities within the context of the overall life cycle. Therefore, in order to confirm the necessity of developing South African specific datasets for end-of-life, it is recommended that the relative importance that making such changes be assessed within the context of a product's life cycle.

Finally, given the potential for future ecoinvent releases to contain improved parametrisation with regards to representing alternative landfill practises and climatic conditions, it is recommended that future versions of the model and corresponding datasets be evaluated and assessed with regards to the extent of their new capabilities. This assessment should not only consider the physical capabilities of models and datasets, but further assess the accessibility of these changes. Improving the accessibility of parameterisation within models and datasets is necessary to allow these capabilities to be understood and applied by LCA practitioners. It is the objective of parameterisation to allow a wider range of conditions to be represented and thus, imposing a parameterisation potential on models and datasets must be done in such a manner that utilising this potential does not fall outside the scope of a product LCA.

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APPENDIX A

SUPPLEMENTARY RESULTS

A.1 National Waste Legislation and Standards

This section provides an overview of waste legislation in South Africa, starting with the *NEM:WA* (2009), and followed by supporting policies and regulations outlining the responsibilities of national, provincial, and municipal governments in implementing and enforcing this legislation. It is not the intention of this overview to review and provide commentary on all relevant waste legislation, but rather to present key pieces that establish the framework for waste management in South Africa. Additional waste legislation, identified on the South African Waste Information Centre (SAWIC) but not discussed in further detail here, are listed in Table A.1.

Table A.1 Key waste legislation in South Africa as reported on the SAWIC (2017b)

Legislation
The South African Constitution (Act 108 of 1996)
Hazardous Substances Act (Act 5 of 1973)
Health Act (Act 63 of 1977)
Environment Conservation Act (Act 73 of 1989)
Occupational Health and Safety Act (Act 85 of 1993)
National Water Act (Act 36 of 1998)
The National Environmental Management Act (Act 107 of 1998)
Municipal Structures Act (Act 117 of 1998)
Municipal Systems Act (Act 32 of 2000)
Mineral and Petroleum Resources Development Act (Act 28 of 2002)
Air Quality Act (Act 39 of 2004)
National Environmental Management: Waste Act, 2008 (Act 59 of 2008)
National Environmental Management: Waste Amendment Act, 2014 (Act 26 0f 2014)

A.1.1 National Environmental Management: Waste Act (Act 59 of 2008)

The *NEM:WA* (2009) was written with the objective of reforming the regulation of waste management in South Africa to provide a more holistic approach to ensure the protection of both human health and the environment (DEA, 2011a). A key requirement of the *NEM:WA* (2009) is the adherence of any person who produces or handles waste to a set of regulatory standards. The internationally recognised waste hierarchy (Figure 1.2) is an underlying principle of the *NEM:WA* (2009), supporting a shift away from traditional treatment such an landfilling, instead promoting resource efficiency (UNEP, 2011).

The *NEM:WA* (2009) lays out the basis for various regulatory and economic instruments to give effect to the *National Environmental Management: Waste Act, No. 59 of 2008. National Waste Management Strategy* (2012) (hereafter, *NWMS*). An overview of the key instruments set out in the *NEM:WA* (2009) (as defined in the *NWMS* (2012:s3, ss3.1) are as follows:

- **Waste classification and management system:** Provides a methodology for the classification of waste and provides standards for the assessment and disposal of waste for landfill disposal.

- **Norms and standards:** Establishes baseline regulatory standards for managing waste at each stage of the waste management hierarchy.
- **Licensing:** Lists activities that require licenses (with conditions) and those that do not if undertaken according to conditions or guidelines.
- **Industry waste management plans:** Enables collective planning by industry to manage their products once they become waste and to collectively set targets for waste reduction, recycling and re-use.
- **Extended producer responsibility (EPR):** Regulates that industry is responsible beyond point of sale for particular products that have toxic constituents or pose waste management challenges, particularly where voluntary measures have failed.
- **Priority wastes:** Identifies categories of waste, that due to their risks to human health and the environment, require special waste management measures, particularly where a solution requires the involvement of multiple role-players.
- **Economic instruments:** Encourages or discourages particular behaviour and augments other regulatory instruments.

The approach of the *NEM:WA* (2009) is not to provide specific details on the management of particular waste types, but rather to develop the framework for waste legislation, and provide enabling powers to the Minister of Environmental Affairs and relevant Members of Provincial Executive Committees (MECs) to promulgate further regulations on necessary aspects thereof (DEA, 2011a).

A.1.2 The National Waste Management Strategy

The *NWMS* (2012) is a legislative requirement of the *NEM:WA* (2009), written with the objective of defining South Africa's priorities for waste management, and identifying challenges and possible solutions with respect to these priorities (DEA, 2011a). The *NWMS* (2012) represents a binding national strategy, requiring the compliance of all national, provincial, and municipal departments, as well as the private sector and general public (DEA, 2011a). It is intended that this strategy be regarded as a "living document" (*NWMS*, 2012:70), and as such, it is stipulated in Section 6(5) of the *NEM:WA* (2009) that the *NWMS* be reviewed and updated at least every five years. The *NWMS* (2012) is structured around a framework of eight goals, each of which have an associated set of targets, which were to be met by 2016. The complete set of associated targets is available in Table A.2 overleaf.

Table A.2 Summary of the National Waste Management Strategy goals and targets (Adapted from the NWMS, 2009:9)

Goal	Target
1. Promote waste minimisation, re-use, recycling and recovery of waste	• 25% of recyclables diverted from landfill sites for re-use, recycling or recovery • All metropolitan municipalities, secondary cities and large towns have initiated separation at source programmes • Achievement of waste reduction and recycling targets set in IndWMPs for paper and packaging, pesticides, lighting (CFLs) and tyres industries
2. Ensure the effective and efficient delivery of waste services	• 95% of urban households and 75% of rural households have access to adequate levels of waste collection services • 80% of waste disposal sites have permits
3. Grow the contribution of the waste sector to the green economy	• 69 000 new jobs created in the waste sector • 2600 additional SMEs and cooperatives participating in waste service delivery and recycling
4. Ensure that people are aware of the impact of waste on their health, well-being and the environment	• 80% of municipalities running local awareness campaigns • 80% of schools implementing waste awareness programs
5. Achieve integrated waste management planning	• All municipalities have integrated their IWMPs with their IDPs, and have met the targets set in IWMPs • All waste management facilities required to report to SAWIS have waste quantification systems that report information to WIS
6. Ensure sound budgeting and financial management for waste services	• All municipalities that provide waste services have conducted full-cost accounting for waste services and have implemented cost-reflective tariffs
7. Provide measures to remediate contaminated land	• Assessment complete for 80% of sites reported to the contaminated land register • Remediation plans approved for 50% of confirmed contaminated sites
8. Establish effective compliance with and enforcement of the Waste Act	• 50% increase in the number of successful enforcement actions against non-compliant activities • 800 EMIs appointed in the three spheres of government to enforce the Waste Act

A.1.3 National Environmental Management: Waste Amendment Act (Act 26 of 2014)

The objective of the *National Environmental Management: Waste Amendment Act, No. 26 of 2014* (2014) is to address the queries and needs arising from the promulgation of the *NEM:WA* (2009), and provide clarity and resolution on various aspects therein. An important amendment is the refinement of what constitutes waste, and hence the revised definition of the materials/substances and activities requiring a waste license. For example, importantly for the country's large minerals industry, residues from this sector, which had previously been excluded, have now been defined as wastes. These revisions are important, as they provide support for the definition of end-of-life criteria, specifying when a waste obtains the status of a product or a raw material, with this definition intended to support opportunities to regulate and support the recycling economy (Frittelli & Govender, 2014).

The *National Environmental Management: Waste Amendment Act, No. 26 of 2014* (2014:s13) further provides for the establishment of a Waste Management Bureau and defines its role in the implementation of waste policy. In terms of Section 13 of the Amendment Act, one of the main objectives of the Bureau is to serve as a "specialist implementing agent" in terms of the *NEM:WA* (2009), and in so doing, promote and facilitate the uptake of the waste management hierarchy, manage the disbursement of incentives and funds for waste management operations, monitor the implementation of integrated waste management plans (IWMPs), build capacity to provide specialist support for the implementation of waste management strategies, and function in an advisory capacity for the development of waste management plans and strategies.

A.1.4 National Pricing Strategy for Waste Management

A further important amendment contained in the *National Environmental Management: Waste Amendment Act, No. 26 of 2014* (2014:s6) is the definition of a national pricing strategy, providing the basis and guiding methodology for setting waste management charges for all waste management practices. The resulting *National Environmental Management: Waste Act, No. 59 of 2008. National Pricing Strategy for Waste Management* (2016) (hereafter, *NPSWM*) is a legislative requirement of the Amendment Act and is aligned with the objectives of the *NEM:WA* (2009). The objective of the *NPSWM* (2016) is to support the implementation of various economic instruments to encourage a change in behaviour towards waste generation and management across all sectors of society. This approach includes encouraging the "polluter pays principle", reducing waste generation, supporting sustainable waste management practices, and supporting the growth of a regional secondary resources economy from waste (*NPWSM*, 2016:s2, ss2.3). Potential economic instruments identified in the *NPSWM* (2016) are varied and can be implemented at various stages along the waste value chain. The possible economic instruments identified in the *NPSWM* (2016:16) are shown in Table A.3.

Table A.3 Potential economic instruments for solid waste management as identified and presented in the National Environmental Management: Waste Act, No. 59 of 2008. National Pricing Strategy for Waste Management (2016:16)

Category	Economic Instrument
Downstream instruments	• Volumetric tariffs ("pay-as-you-throw") • Waste disposal taxes
Upstream instruments	• Material and input taxes • Product taxes • Advance recycling fees • Deposit-refund schemes • EPR schemes
Subsidy-based instruments	• Recycling subsidies • Tax rebates and benefits • Capital financing

A.1.5 National Domestic Waste Collection Standards

Of particular relevance to the management of municipal waste are the *National Environmental Management: Waste Act, No 59 of 2008. National Domestic Waste Collection Standards* (2011), also a legislative requirement of the *NEM:WA* (2009) (Section 7(1)(b)). The purpose of these Standards is to address the historical lack of waste collection standards in the country, as well as the imbalance in the provision of waste collection services. The provision of waste collection services is integral in maintaining acceptable living and working conditions and, as such, are also informed by the *The Constitution of the Republic of South Africa, Act No. 108 of 1996* (1996), giving effect to the right of citizens, under Section 24(a), to an "environment that is not harmful to their health or well-being".

It is stipulated within the *NEM:WA* (2009:4) that standards are required to promote and give effect to this right, and that this right be applied "uniformly throughout the Republic". However, in recognition of South Africa's status as a developing country, Section 3 of the resulting Standards makes provision for a variation in the service level between areas, with the provision of services influenced by the "practicality and cost efficiency of delivering the service". For example, according to Section 3 and Section 4 of these Standards, rural areas such as farms and low-density settlements may be provided with appropriate, on-site disposal under the regular supervision of a waste management officer. Medium density settlements may utilise a community transfer to a central collection point, while high density settlements require organised transfer to a centralised collection point and/or kerbside collection.

According to Section 4(4.2) of these Standards, municipalities must provide an "enabling environment" for households to recycle waste. Separation at source is an important factor in creating such an environment and, as such, Section 4(4.1) lays out the framework for its implementation, stipulating that all domestic waste be separated at source in all metropolitan and secondary cities. According to Section 4(4.2), where the municipality does not provide for the kerbside collection of the source separated material, it must make provision for a well-functioning drop-off centre within close proximity, where recyclables can be dropped off by households or the relevant service provider.

A.1.6 National Waste Information Regulations

The *National Environmental Management: Waste Act, No. 59 of 2008. National Waste Information Regulations* (2012) was developed with the objective of regulating the collection and reporting of waste data to the SAWIS, as per Section 60 of the NEM:WA (2009). Under these Regulations (Chapter 2, Section 5) it is stipulated that all listed waste management activities be registered on the SAWIS within 90 days of the promulgation of the Regulations. It is further specified in Section 6(1), that any listed activities commencing after the promulgation of the Regulations be registered on the SAWIS within 30 days of their commencement.

This suggests that the SAWIS should provide a comprehensive source of waste data for South Africa. Public access to the waste data reported to the SAWIS is available online through the SAWIC (DEA, 2017b). This data is available as tonnage reports and can be filtered by type of waste activity for municipality, province, Standard Industrial Classification (SIC) code, waste type, and management option. The categorisation of waste management options on the SAWIS is in accordance with the categorisation of waste management defined in Annexure 5 of the *National Environmental Management: Waste Act, No. 59 of 2008. National Waste Information Regulations* (2012). This categorisation is only strictly applicable to waste management facilities required to register on SAWIS in accordance with Section 5 of the Regulations. These criteria are available in Table A.4 overleaf.

Table A.4 List of persons required to register on the SAWIS in terms of the National Waste Information Regulations National Environmental Management: Waste Act, No. 59 of 2008. National Waste Information Regulations (2012:s5)

Activity	Conditions for Registration
Generators of Waste	• Generators of hazardous waste in excess of 20 kg per day
Disposal of Waste	• Disposal of general waste to land covering an area in excess of 200m^2 • Disposal of any quantity of hazardous waste to land
Treatment of Waste	• Treatment of general waste using any form of treatment at a facility that has the capacity to process in excess of 10 tonnes of general waste or 500 kg of hazardous waste per day excluding the treatment of effluent, wastewater or sewerage • Treatment of health care risk waste regardless of size or capacity of the facility
Recovery of Recycling of Waste	• Recovery of energy from general waste in excess of three tonnes per day • Recovery of waste at a facility that has the capacity to process in excess of 10 tonnes of general waste or in excess of 500 kg of hazardous waste per day, excluding recovery that takes place as an integral part of an internal manufacturing process within the same premises • The scrapping or recovery of motor vehicles at a facility that has an operational area in excess of 500 m^2 • Recycling of general waste at a facility that has an operational area in excess of 500 m^2 • Recycling of hazardous waste in excess of 500 kg per day calculated as a monthly average
Exportation of hazardous waste	• Hazardous waste exported from the Republic of South Africa

A.1.7 Distribution of Responsibility for Waste Management Amongst Government Structures

The government of South Africa is comprised of three spheres: national, provincial, and local (municipal) which, according to *The Constitution of the Republic of South Africa, Act No. 108 of 1996* (1996:chap 3, s40), are at once “distinctive, interdependent and interrelated”. All three spheres are attributed with both legislative and executive authority (du Plessis & du Plessis, 2011)[20]. In terms of waste management, the relevant legislation, strategies, plans, and policies are set by all three spheres of government. The *NEM:WA* (2009) provides the framework for waste management and specifies the role of the national government in meeting the objectives outlined for sustainable waste management. In terms of the *NEM:WA* (2009), it is the duty of the Minister to develop and implement specific policies, guidelines and standards in alignment with the objectives of the Act (Odeku, 2014).

Section 7(2) of the *NEM:WA* (2009) makes provision for the regionalisation of waste management services, with the responsibilities of the relevant provincial MECs laid out in Section 8. According to the *NEM:WA* (2009:s8), it is the responsibility of the relevant MEC to not only ensure that the national waste management strategy and national norms and standards are implemented within their jurisdiction (Section 8(1)), but to set provincial norms and standards that are not in conflict with national norms and standards (Section 8(2)). It is specified in the *NEM:WA* (2009:s8, ss3) that these provincial norms and standards must facilitate and advance, amongst else, provincial planning and provision of waste

[20] This authority is defined for national, provincial and local government in terms of Sections 44, 104 and 156 respectively in *The Constitution of the Republic of South Africa, Act No. 108 of 1996* (1996).

management services, the development of provincial re-use, recycling, recovery and waste minimisation strategies, and plans for the regionalisation of waste management services.

The role of the municipality is defined in Section 9 of the *NEM:WA* (2009). According to Section 9(1), the principle authority of the municipality is to “deliver waste management services, including waste removal, waste storage and waste disposal services, in a manner that does not conflict with Section 7 or 8 of [the] Act”. The executive authority of the municipality and corresponding duties in relation to waste services are laid out in Section 9(2) of the *NEM:WA* (2009). These include the development, implementation, and integration of municipal IWMPs with national and provincial plans, and ensuring equitable and affordable access for all to waste management services. Section 4(3) makes provision for the Minister or relevant MEC to act in terms of the *NEM:WA* (2009) in relation to a municipality to “support and strengthen the municipality’s ability or right to perform its functions in relation to waste management activities”. This provision therefore allows the national and provincial government to intervene into municipal duties, should the municipality be incapable of meeting its waste management obligations (Odeku, 2014).

While each sphere of government has a defined responsibility in terms of waste management, these functions are not fully isolated, and thus there is a high level of interdependence and interaction between the spheres. This can be illustrated by the requirements for the development of IWMPs. Municipal and provincial IWMPs provide a strategy for appropriate waste collection standards and a target for how they can be achieved. It is required that municipal IWMPs are ratified by the provincial government. These then inform the development of provincial IWMPs, which are submitted to the national government for approval and to ensure alignment with the national strategy (DEA, 2011a). However, in practice, the need for alignment between municipal and provincial plans and strategies has resulted in inconsistencies in waste planning and reporting, with various aspects of waste management being addressed in different documents in different provinces.

A.2 Supplementary SAWIS Data

Table A.5 Tonnage report showing total waste disposal by waste type and management option as reported to the SAWIS and obtained from the SAWIC tonnage reports (DEA, 2017b) (last updated 20 July 2017)

Waste Type	SAWIS Tonnages by Management Option ('000 tonnes)			
	Disposal	Treatment	Waste Recovery & Recycling	Hazardous Waste Exporter
GW01 Municipal Waste	10902	103.14	152.25	
GW10 Commercial and Industrial Waste	1351.9	4.0097	24.059	
GW13 Brine	6.1977			
GW14 Fly ash and dust from miscellaneous filter sources	1.0920			
GW15 Bottom Ash	16482		20.063	
GW16 Slag			0.0038	
01 Ferrous metal slag	1534.1		230.88	
02 Non-ferrous metal slag	221.69		26.741	
03 Slag: Other	271.02		233.95	
GW17 Mineral waste	4.9357		0.0188	
01 Foundry sand	2.5668			
02 Refractory waste	4.7591		12.095	
03 Other	268.49		0.6658	
GW18 Waste of Electric and Electronic Equipment (WEEE)	0.0132	10.743	21.494	
03 Office, information and Communication Equipment			0.3640	
05 Lighting equipment			95.847	
06 Electric and electronic tools			0.0109	
08 Mixed WEEE			48.882	
GW20 Organic waste	914.76	20.675	14.026	
01 Garden waste	301.30	142.28	32.833	
02 Food waste	27.474	12.947	38.028	
03 Wood waste	1019.8	13.231	305.84	
GW21 Sewage sludge	4.6163		0.1762	
GW30 Construction and demolition waste	1841.8		236.10	
GW50 Paper	0.0220	11.169	1043.8	
01 Newsprint and magazines	0.2320		154.42	
02 Brown grades		49.954	1173.7	0.9477
03 White grades			353.44	
04 Paper: Mixed grades		4.8476	312.93	
GW51 Plastic	10.004		208.58	
01 Polyethylene terephthalate	1.3118		26.376	
02 Polyvinylchloride			4.0391	
03 Low-density Polyethylene			211.27	
04 Polypropylene			11.442	
05 Polystyrene	0.0117		7.1882	
06 Other			29.881	

Continuation of Table A.5 Tonnage report showing total waste disposal by waste type and management option as reported to the SAWIS and obtained from the SAWIC tonnage reports (DEA, 2017b) (last updated 20 July 2017)

Waste Type	**SAWIS Tonnages by Management Option ('000 tonnes)**			
	Disposal	**Treatment**	**Waste Recovery & Recycling**	**Hazardous Waste Exporter**
High-density Polyethylene*			81.524	
GW52 Glass	0.2095		154.98	
GW53 Metals	0.0336		93.378	
01 Ferrous metal	0.0925		12158	
02 Non-ferrous metal	0.0012		331.29	
GW54 Tyres	0.5330	8.3016	4.4996	
GW99 Other	599.29	0.5031	31.51	

A.3 Supplementary SAWIS Analysis

An illustration of the current limitations of the SAWIS as a source of representative waste data is available in Table A.6. This provides a comparison of recycling and recovery data obtained from the SAWIS with recovery data obtained from the *GreenCape Waste Economy Market Intelligence Report* (GreenCape, 2016) and packaging material tonnages provided by Packaging SA.

Table A.6 Comparison of recycling and recovery data reported to the SAWIS for common packaging materials with national recovery information obtained from alternative sources

Material	**SAWIS[a] (DEA, 2017b)**	**GreenCape Waste Economy: Market Intelligence Report[b]**	**Packaging SA Packaging Material Tonnages[c]**
Description of reported value	Material reported under Waste Recycling and Recovery[d]	Material diverted from landfill in 2016	Packaging material collected in 2015
	Tonnes		
Paper	3 038 000	1 100 000	1 196 000
Plastic	580 000	315 000	365 200
Glass	155 000	338 000	278 600
e-Waste	167 000	45 000	-
Metal	12 580 000	2 497 000[e]	152 300
Tyres	4 500	109 900	-

[a] Data generated using SAWIC tonnage reports (SAWIC, 2017b). Last updated 20 July 2017

[b] Source: GreenCape (2016). Information obtained from Producer Responsibility Organisations (PROs) representing the particular material. Data representative of the national level

[c] Source: Data copyright to Packaging SA (provided by BMI Research (2016)). Reproduced here with written permission from Packaging SA

[d] Sum of all material reported to all general waste categories representing the particular material

[e] Reported as scrap metal

While the three data sources shown in in Table A.6 are not directly comparable due to the differences in the definition of recovery used and the representative year, comparison thereof is useful in gauging perspective on the extent to which the SAWIS can be considered representative of the sector. The SAWIS data explicitly represents waste management activities reported under "Recycling and Recovery" with alternative treatment reported under separate activities. It is anticipated that the resultant quantities should be consistently lower than those reported by GreenCape (2016), who report on *total* material diverted from landfill, thus including both recycling and alternative treatment. Although Figure 4.2 suggests that the contribution of alternative treatment towards general waste management is relatively low, Table A.6 shows that the SAWIS data is notably higher than that reported by

GreenCape (2016) for various categories. The most notable discrepancy occurs for metal, with the SAWIS system reporting a value 10 million tonnes higher than that reported by GreenCape (2016). This discrepancy appears to arise from the quantity of ferrous metals reported to the SAWIS (12 158 000 as compared to 331 000 tonnes for non-ferrous metals), suggesting an error in reporting. The quantity of e-Waste reported by the SAWIS is also notably higher than that reported by GreenCape (2016).

Even where the SAWIS quantity is lower than the GreenCape (2016) alternative, this result appears questionable. For example, the quantity of glass recycled according to the SAWIS is lower than that reported by Packaging SA. Packaging SA report only on the recovery of glass packaging tonnages, which intuitively should be lower than a total glass quantity. A possible explanation for the lower quantities reported by the SAWIS could lie in the waste categorisation used by the system, where categories such as GW01 and GW10 contain additional material not counted in Table A.6. However, if notable quantities of the recyclables listed in Table A.6 are contained in GW01 and GW10, then it would be expected that the SAWIS data be consistently lower than that reported by GreenCape (2016). This, however, is not observed, suggesting that the discrepancy must be attributable to other factors.

Further limitations in the SAWIS data can be observed through a comparison of this data with that reported in the *NWBR* (DEA, 2012b). This comparison is shown in Table A.7. Notable discrepancies occur in the quantity of metal (GW53) reported to the SAWIS in addition to the total quantity of tyres (GW54). With regards to the latter, the recovery of tyres was an explicit mandate of REDISA, and thus the limited growth in the reported quantity of recovered tyres, when compared to that reported in the NWBR, suggests that this quantity has been under-reported.

Table A.7 Comparison of the tonnages of waste reported to the SAWIS and by the NWBR (DEA, 2012a) for the management of the non-landfilled portion of general waste

Waste Type	**SAWIS Data**		**NWBR Data**
	Treatment	**Waste Recovery and Recycling**	**Recycling**
	Kilo Tonnes		
GW50 Paper	65.97	3038.3	988.6
GW51 Plastic	-	580.3	235.5
GW52 Glass	-	155.0	307.1
GW53 Metals	-	12580	2497
GW54 Tyres	8.302	4.500	9.865

A.4 South African Waste Characterisation Studies

While waste generation and disposal data enables the determination of the status quo with regards to the total quantity of waste produced and managed via different options, characterisation data provides a level of depth to this analysis, providing information on a specific waste type and further informing the potential for alternative management thereof. However, waste characterisation studies undertaken in South Africa at both national and municipal level are limited (Oelofse, Muswema & Koen, 2016). From a municipal perspective, it has been claimed that larger municipalities tend to provide some level of waste composition data, albeit with varying frequencies, while smaller municipalities tend to lack any form of data at all (Friedrich, 2013). While this observation might be true for the general case, it is likely that smaller well-run municipalities — such as Hessequa or Eden for example — have good data records. Furthermore, given the influence of income on both waste generation and composition, the "single figure" reporting approach adopted by a number of municipalities, in which income groups are combined, represents a shortcoming in the accuracy and applicability of data generated (Oelofse, Muswema & Koen, 2016:346). Thus, not only is waste characterisation data constrained by availability, but the inconsistencies in sampling methodologies and reporting style between municipalities imposes an additional constraint on the use of this data for comparative purposes (DEA, 2012a).

A comprehensive list of waste characterisation studies conducted for different South African municipalities is presented in the Appendix to the *NWBR* (DEA, 2012b). This list is by no means exhaustive, but rather provides an overview of characterisation studies where a clear and consistent distinction has been made between different socio-economic groups. Comparison of the results obtained from these studies supports the observations of Oelofse, Muswema and Koen (2016), reflecting significant variation in both the categorisation of waste used and the reported waste fractions. These discrepancies have been attributed to the lack of standardisation of waste characterisation methodology, resulting in the definition and use of unrepresentative sample sizes, and an insufficient sampling period for capturing seasonal variation in waste composition and generation (DEA, 2012b).

Where characterisation studies are representative of a particular waste fraction over a limited period — such as residential waste within a particular area — these are referred to as a "micro-characterisation" studies (Western Cape Government Department of Environmental Affairs and Development Planning [WCDEADP] (2010) as cited by Myers and Pieterse (2014:11)). Developing countries typically exhibit a high variance in waste characterisation results, due to variations in factors such as income levels, domestic fuel supply, and living standards (Troschinetz & Mihelcic, 2009). Given that micro-characterisation studies represent only a fraction of waste, to represent waste composition within the national context a "macro-characterisation" is required, involving an on-going quantification and characterisation of all waste entering disposal sites (WCDEADP (2010), as cited by Myers and Pieterse (2014:11)). While macro-characterisation represents waste composition within a broader context, given that this information is typically obtained from waste disposal sites, waste sources tend to be aggregated, and hence information can be lost regarding socio-economic characterisation.

A detailed summary of macro-characterisation waste studies undertaken either by, or on behalf of, nine South African municipalities has been presented by Friedrich (2013). These results provide an indication of the composition of MSW collected and reaching landfill sites on a national scale. The weighted average result of this summary is shown in Table A.8 overleaf.

Table A.8 Weighted average South African municipal waste composition collected and reaching landfill sites (Adapted from Friedrich, 2013:4-15)

Waste Component	Weighted Average Waste Composition (% wet waste)
Food waste (Organic)[a]	26.0 ± 2.6
Garden waste (Green)[a]	18.2 ± 1.14
Glass	6.9 ± 0.54
Metals	3.9 ± 0.35
Paper	18.2 ± 0.65
Plastic	12.1 ± 0.45
Other	14.7 ± 3.35

[a] Where food and garden waste were classified together, a 50/50 proportion was assigned to each.

While Table A.8 provides an indication of the national waste composition reaching landfills in South Africa, the variation in waste composition with region and income group shown in the Appendix to the *NWBR* (DEA, 2012b) suggests that the use of this composition to inform decision-making regarding waste management is limited. According to Oelofse, Muswema and Koen (2016), accurate information on both composition and volume by waste stream, and geographic area are critical requirements for waste management planning, and thus findings based on aggregated waste data might not be applicable to all regions and communities within South Africa. For example, the implementation of source-separation programs to recover recyclable material might be feasible in higher income areas with a high proportion of recyclable packaging in the waste stream, but less feasible in low income areas with a higher proportion of organic waste. Such challenges facing waste management planning are not unique to South Africa, and it has been suggested that limitations in a standard methodology for waste characterisation studies exists at both national and international levels (Dahlén & Lagerkvist, 2008).

A.5 Overview of Modelling Approach for the Estimation of Informal Household Waste

The following section provides an overview of the modelling approach used to estimate informal domestic waste generation in South Africa. It should be noted that this section is intended to be read in conjunction with Section 4.4.1. Certain details regarding modelling assumptions are not repeated and are as reported in Section 4.4.1.

The first step in the modelling approach required the determination of the population per income group and settlement type. This was undertaken using the data shown in Table 4.5, as according to Equation A.1. The results showing the population distribution per income group is available in Table A.9.

$$\text{Population}_{\text{settlement type, income group}} = 54\ 978\ 907 \times \%\ \text{population}_{\text{settlement type, income group}}$$

Equation A.1

Table A.9 2016 South African population distribution

Population Distribution[a]	**No Income**	**Low Income**	**Middle Income**	**Upper Income**
Rural Population	2 547 997	7 461 737	7 207 955	305 023
Urban Population	5 973 733	8 482 146	19 291 879	3 708 437
Total	**8 521 731**	**15 943 883**	**26 499 833**	**4 013 460**

[a]Total population based on 2016 population estimate reported by [1]Worldometers (2017) and population income distribution based on household income distribution reported by Stats SA (2015)

The next step in the approach required the determination of total domestic waste generation per income group and settlement type. This was undertaken using the data shown in Table 4.5 and Table A.9. in Equation A.2. This equation was applied for each income group for rural and urban settlements. The results showing the waste generation per income group is available in Table 4.6.

$$\text{Domestic waste}_{\text{settlement type, income group}} = \text{WGR}_{\text{income group}} \times \text{population}_{\text{settlement type, income group}}$$

Equation A.2

The final step in the approach required the estimation of the informal proportion of waste per income group and settlement type. This was based on the distribution of waste services as reported in the *General Household Survey 2015* (Stats SA, 2016a) in Table 4.4. It was assumed that any form of regular waste collection service and centralised communal dumps constitute a formal waste management service, with the last three categories shown in Table 4.4 constituting informal waste management. Informal domestic waste generation per income group and settlement type was then calculated according to Equation A.3. Further assumptions regarding the distribution of waste services are as reported in Section 4.4.1. Equation A.3 was applied to each income group for rural and urban settlements. The results showing the waste generation per income group is available in Table 4.6.

$$\text{Informal domestic waste}_{\text{settlement type, income group}} =$$

$$\text{Domestic waste}_{\text{settlement type, income group}} \times \%\ \text{un-serviced household}_{\text{settlement type, income group}}$$

Equation A.3

A.6 Distribution of General Waste Between Different Landfill Classes

To date, no data is available providing the actual flow of waste to each landfill site operating in South Africa, and the determination of a suitable range in the relative contribution from each size class requires an alternative approach. This estimate was developed from a list of licensed landfill operations in the country (available from the SAWIC (DEA, 2017b)), which was screened to isolate sites accepting general waste. The resulting list was then sorted according to landfill size classification. This list provided a record of licensed facilities, but did not contain any information regarding daily waste flows received by each site. The relative quantity of general waste treated in each size class was estimated using the assumptions outlined in Section 4.6.2. A summary of the key data used in this estimate and corresponding results are shown in Table A.10.

Table A.10 Summary of key data and results for the estimate of the distribution of general waste between landfill classes

Size Class	Expected Daily Waste Flow[a] (tons/day)	Number of Sites[b]	Approximate Daily Flows per Size Class (tons/day)	Waste Distribution (% of Total Flow)	B^- Sites (% per Size Class)[c]	% of Licensed Sites per Size Class
C: Communal	8	426	3 408	2.8	90	50
S: Small	66	224	14 784	12	84	27
M: Medium	273	139	37 947	31	70	16
L: Large	1345	49	65 905	54	47	5.8

[a]Assuming a lognormal probability distribution function
[b]Licensed general waste disposal facilities following the screening of disposal facilities in the list available on the SAWIC (DEA, 2017b)
[c]Climatic water balance classification provided in the list available on the SAWIC (DEA, 2017b)

In order to establish an expected range for this data, the daily deposition rate was varied between the lower and upper limits specified for each landfill size class. This approach provided a possible range with regards to the relative contribution from each size class to managing South Africa's landfilled waste. These results are illustrated in Table A.11.

Table A.11 Estimation of a possible range in the distribution of general waste between landfill classes in South Africa

Size Class	Average Daily Flow per Site (tons/day)	% Contribution using Lower Limits	% Contribution using Upper Limit
C: Communal	2.5[a] – 25	2.0	3.0
S: Small	25 – 150	12	9.6
M: Medium	150 – 500	38	18
L: Large	500 - 5000	50	70
Annual Flow$_{\text{Lower limits}}$	**17 830 000**		
Annual Flow$_{\text{Upper limits}}$	**128 000 000**		

[a] A lower limit for this class was assumed to be 10% of the maximum flow

Although the sensitivity analysis shown in Table A.11 provides a possible range for the total quantity of landfilled waste in South Africa, the relative proportion of waste disposed of in each site towards this total need not necessarily fall within the specified range. For example, if the majority of small landfills accept the upper limit of their daily waste flows, and medium and large sites accept the lower limit, the relative proportion of the contribution from these sites will change. However, in lieu of data specifying the average daily waste flows within each size class, the range shown in Table A.11 was used as an indication of the possible contribution from each size class towards the management of landfilled waste in South Africa.

A.7 Modification to the Ecoinvent Landfill Emission Model to Incorporate a MCF

The following outlines the proposed modification to the ecoinvent landfill emission model to incorporate the MCF into the specification of LFG emissions.

The existing model approach determines carbon speciation based on the composition of LFG specified by Doka (2003d) for a well-managed anaerobic landfill site (56% CH_4 and 44% CO_2). To account for the effect of the MCF, it is necessary to modify the base composition of LFG that is used when determining the carbon speciation. This approach assumes that the default composition used as a basis for speciation corresponds to a completely anaerobic environment (MCF = 1). It was further assumed that the relative mass of CH_4 and CO_2 would change depending on the degradation environment but the total amount of elemental carbon contained in the waste would be constant.

The base composition for CH_4 and CO_2 in raw LFG is specified in the ecoinvent sanitary landfill model in cells B108 and B109, on sheet "air and energy" in the ecoinvent landfill emission model Excel spreadsheet (13_MSWLFv2.xls)[21]. The modifications outlined below must be made to these cells.

Modification to cell B108 (composition of CH_4):

$$\left(\frac{g_{CH_4}}{m^3_{LFG}}\right)_{\text{modified}} = \left(\frac{g_{CH_4}}{m^3_{LFG}}\right)_{\text{anaerobic landfill site}} \times MCF$$

Equation A.4

$$= \frac{335.7 g_{CH_4}}{m^3_{LFG}} \times MCF$$

Modification to cell B109 (composition of CO_2):

$$\left(\frac{g_{CO_2}}{m^3_{LFG}}\right)_{\text{modified}} = \left(\frac{g_{CO_2}}{m^3_{LFG}}\right)_{\text{anaerobic landfill site}} + \left(\frac{g_{CO_2}}{m^3_{LFG}}\right)_{\text{converted from methane}}$$

Equation A.5

$$= \frac{726.8 g_{CO_2}}{m^3_{LFG}} + \left(\left(\frac{335.7 g_{CH_4}}{m^3_{LFG}} \times (1\text{-}MCF)\right) \times \left(\frac{12 g_{carbon}}{16 g_{CH_4}}\right) \times \left(\frac{44 g_{CO_2}}{12 g_{carbon}}\right)\right)$$

21 Spreadsheet model is available in the ecoinvent database (login to the database possible through www.ecoinvent.org)

A.8 Proposed Adaptation of Waste Degradability with Climate

The following section provides a summary of the approach proposed by Doka (2016) to incorporate the effect of climate on waste degradability.

To account for the effect that precipitation has on landfill emissions, Doka (2016) derived a generic dependence of the extent of decay on precipitation, surmising zero decay for precipitation levels equal to zero and maximum decay equal to and not exceeding that occurring in a temperate climate (precipitation < 1000 mm/year). The proposed adjustment of the decay amount with annual precipitation as presented in Doka (2016:5) is shown in Equation A.6.

$$L0' = L0_t . (1 - e^{-fp.\left(\frac{MAP}{MAP_t}\right)})$$

Equation A.6

Where:

L0' = Adjusted methanogenic decay amount for specified climate

$L0_t$ = Methanogenic decay amount in temperate climate, constant = 60 kg CH_4/t waste

fp = Factor for precipitation-dependence of decay amount, constant = 29.372 [-]

MAP = Mean annual precipitation on landfill site [mm/year]

MAP_t = Mean annual precipitation in temperate climates, constant = 1000 mm/year

A similar adjustment for L0 with temperature is challenged by the scarcity of literature references describing this dependency (Doka, 2016). In lieu of suitable literature, a "coarse" model for this dependency has been proposed by Doka (2016:6) in which it is assumed that maximum decay occurs under temperate climate conditions, with any excess temperature unlikely to increase the extent of decay. This assumption is based on the observation that decomposition heat released in the landfill body typically contributes to the internal temperatures of a sanitary landfill, which can reach up to 120ºC. This implies that elevated atmospheric temperature is unlikely to have any further enlarging effect (Doka, 2016). The decomposition heat generated within the landfill body is further assumed to counteract any significant atmospheric cooling, with the exception of permafrost conditions (Doka, 2016). Under permafrost conditions, it is assumed that the deposited waste is likely to be frozen on delivery, with limited potential to heat up in the ground. This means that the potential for microbial activity promoting decay is limited (Doka, 2016). From an analysis of permafrost occurrence with mean annual temperatures, it was assumed that for temperatures < -2ºC, landfill decay is likely to be affected and that at temperatures < -15ºC, decay will cease (Doka, 2016:7). Following these assumptions, the proposed adjustment of decay amount with mean annual temperature, as presented in Doka (2016:7), is shown in Equation A.7.

$$L0' = L0_t . (1 - e^{-ft.(MAT - 15°)})$$

Equation A.7

Where:

ft = Factor for temperature-dependence of decay amount, constant = 0.3 [-]

MAT = Mean annual temperature on landfill site [mm/year]

Both Equation A.6 and Equation A.7 enable a climate based adjustment for the total decay amount of average waste. However, the ecoinvent sanitary landfill model is structured to determine waste-specific emissions as opposed to average waste behaviour. Given that the emission output from this model depends on waste-specific degradabilities derived for temperate climates, a climate based adjustment for waste-specific degradabilities has been proposed by Doka (2016). This approach uses the dependencies developed in Equation A.6 and Equation A.7 as a basis, adjusted by means of a function for which the following boundary conditions are defined:

- Adjusted degradabilities must remain within the bracket of 0 – 100% so as to ensure mass conservation.
- The ranking of adjusted degradabilities remains consistent regardless of climatic conditions (i.e. a material A that degrades better than a material B in temperate climates will retain this characteristic under different climatic conditions).
- If $L0_t = L0'$ (i.e. temperate climatic conditions) then the adjusted degradability (D') must be equal to the original degradability (D_0).
- If L0' is zero then D' must be zero.

(Boundary conditions adapted from Doka (2016:8))

The corresponding function fulfilling these conditions as presented by Doka (2016:8) is shown in Equation A.8.

$$D' = 1 - e^{\alpha . \ln(1-D_0)}$$

Equation A.8

Where:

D' = Climate-adjusted degradability within 100 years in landfill

D_0 = Original degradability in temperate climate within 100 years in landfill

α = Factor to consistently adjust degradability to a certain climate

The function defined in Equation A.8 is intended to adjust any collection of D_0 values into a corresponding set of D' values with the parameter α used to adjust degradability (Doka, 2016). It can be observed from Equation A.8. that for $\alpha = 1$, $D' = D_0$ and for $\alpha = 0$, $D' = 0$, thus the range $0 < \alpha < 1$ depicts the reduced decay amounts in dry and cold climates relative to their temperate decay (Doka, 2016). Although for $\alpha = \infty$, $D' = 100\%$, it was assumed that the effect of increased precipitation and temperature were unlikely to increase degradation rates beyond those occurring in a temperate climate. The range $0 < \alpha < 1$ is therefore identified to be of primary interest for the purposes of the model (Doka, 2016).

Having developed the requisite function to adjust D_0 values to a corresponding set of D' values, the second stage of the approach develops a climate-dependent α value. Doka (2016) notes that the methanogenic decay amounts (L0) can be converted to an overall degradability by considering firstly, how much carbon has been converted to CH_4 and CO_2 and secondly, how much carbon is available. It has been suggested that if all carbon in an average sample of municipal solid waste were to decay, this would yield a CH_4 output of approximately 220 kg CH_4 per ton of waste (Doka, 2016). Using this estimate, Doka (2016:9) proposes the following relationship to convert L0' values into a value for α :

$$\alpha_x = \frac{\ln\left(1 - \frac{L0'}{L0_{max}}\right)}{\ln\left(1 - \frac{L0_t}{L0_{max}}\right)}$$

Equation A.9

$L0'$ = Adjusted methanogenic decay amount for specified climate

$L0_t$ = Methanogenic decay amount in temperate climate, constant = 60 kg CH_4/ton waste

$L0_{max}$ = Maximal methanogenic decay amount possible, constant 220 kg CH_4/ton waste

Adjusted methanogenic decay amounts can be determined for both precipitation and temperature (Equation A.6 and Equation A.7) and thus, two α values can be determined from Equation A.9, reflecting either a precipitation (α_p) or temperature (α_t) adjustment depending on the choice of L0'. In order to incorporate both adjustments into the α value used in Equation A.7, Doka (2016:9) proposes the multiplication of each individual α value as shown in Equation A.10.

$$\alpha = \alpha_p . \alpha_t$$

Equation A.10

A.9 Supplementary LCIA Results

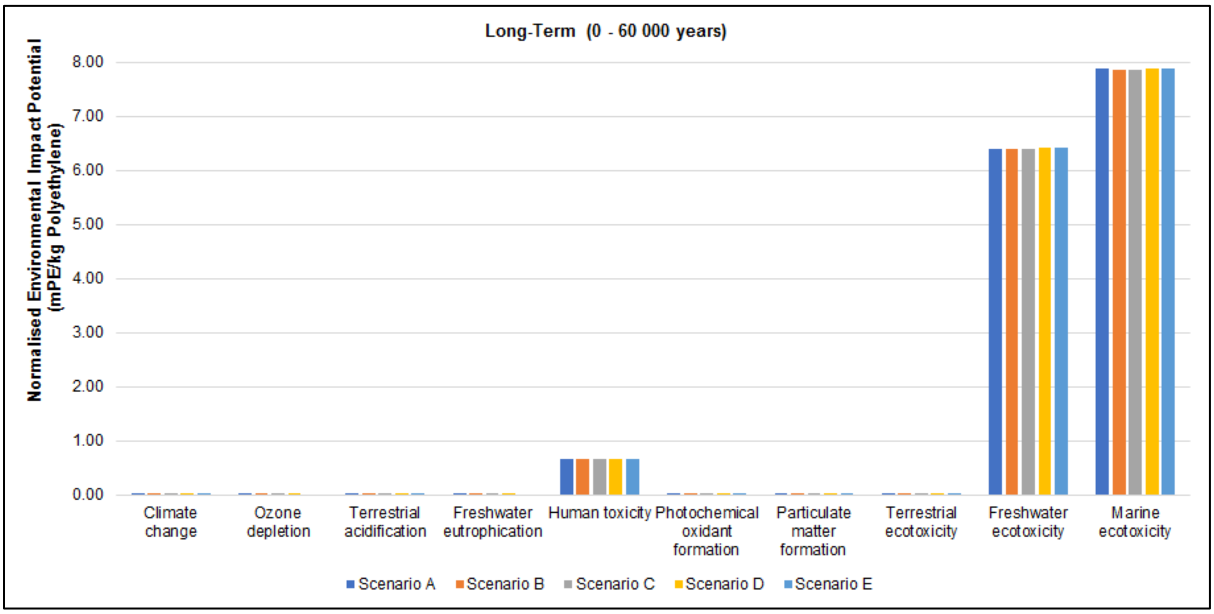

Figure A.1 LCIA results for the treatment of 1kg of polyethylene in the five landfill scenarios investigated. Results are shown as normalised potential impacts for a long-term (0 – 60 000 years) time horizon

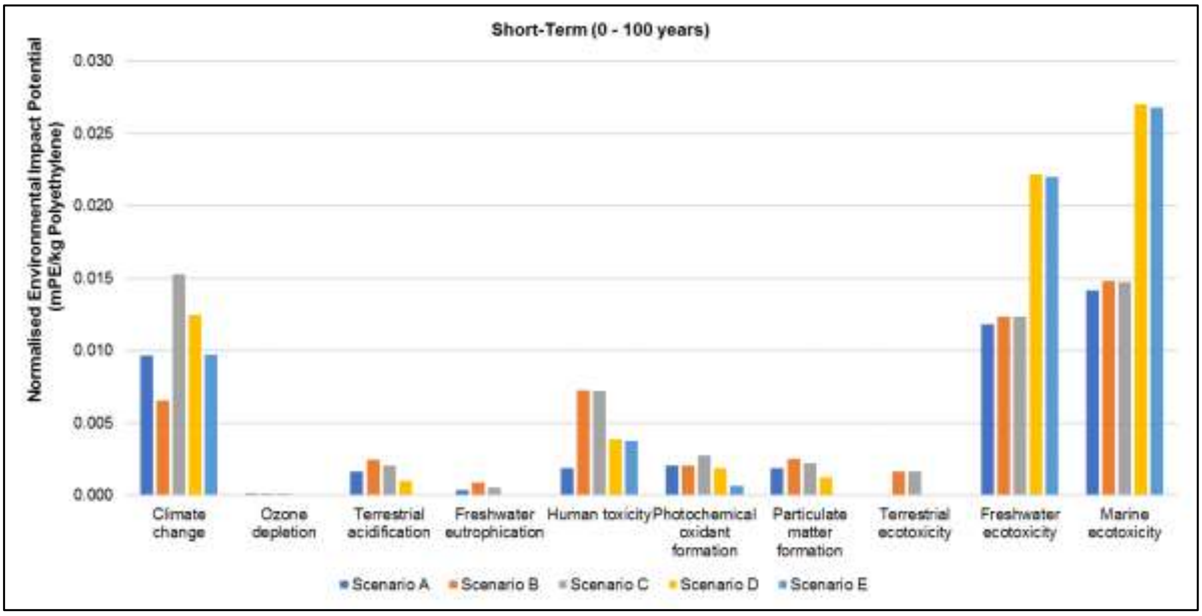

Figure A.2 LCIA results for the treatment of 1kg of polyethylene in the five landfill scenarios investigated. Results are shown as normalised potential impacts for a short-term (0 – 100 years) time horizon

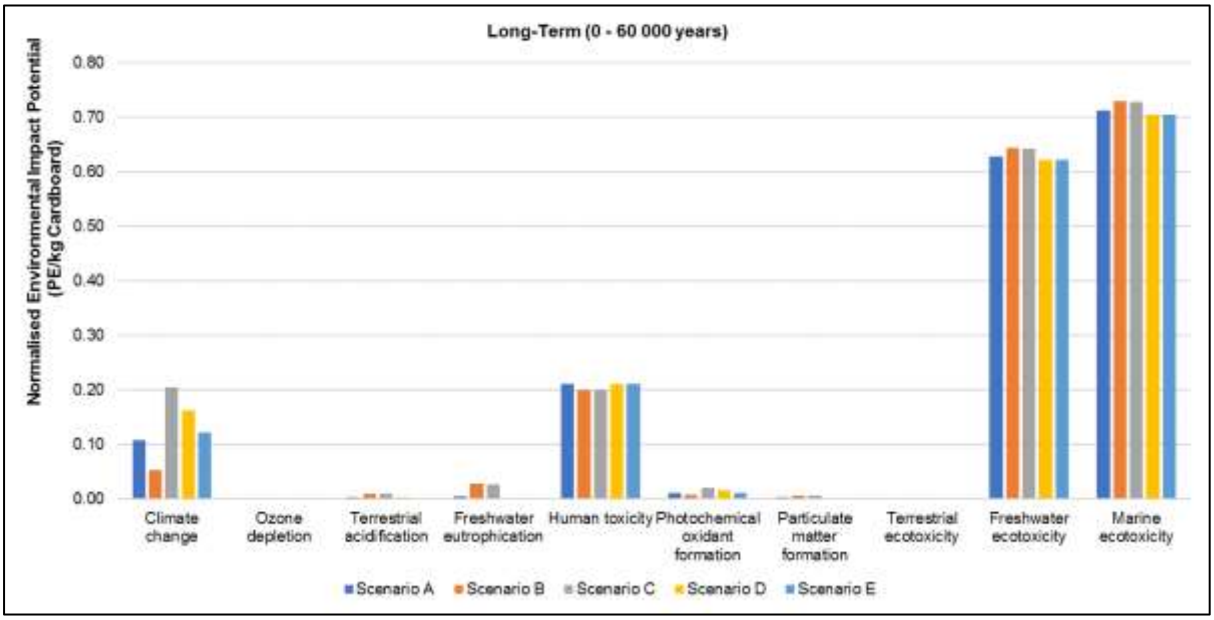

Figure A.3 LCIA results for the treatment of 1kg of cardboard in the five landfill scenarios investigated. Results are shown as normalised potential impacts for a long-term (0 – 60 000 years) time horizon

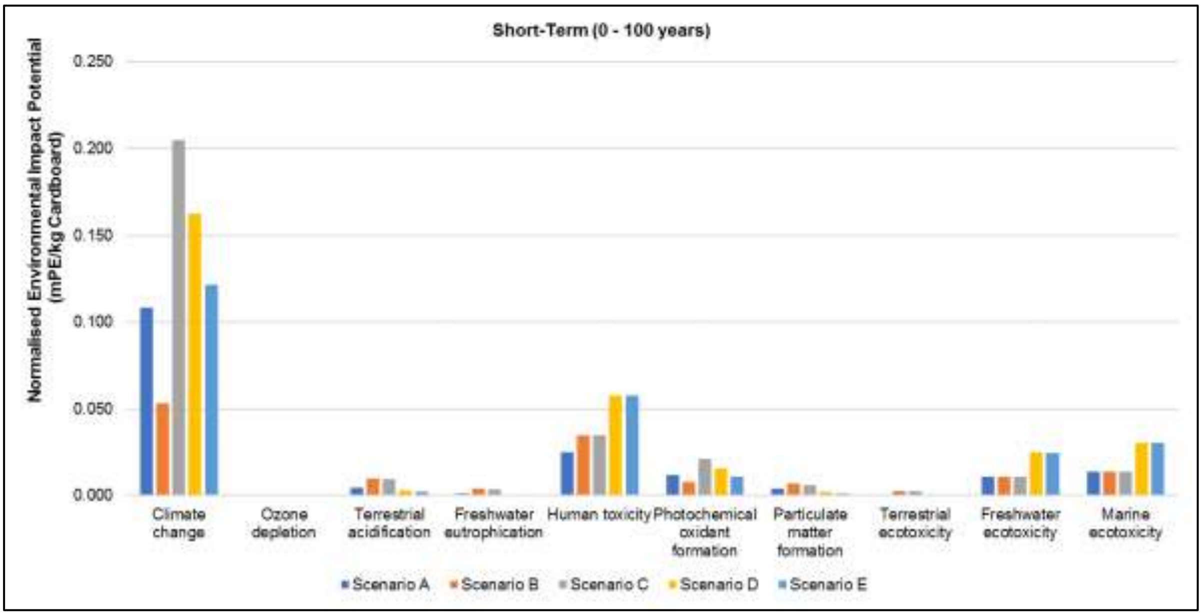

Figure A.4 LCIA results for the treatment of 1kg of cardboard in the five landfill scenarios investigated. Results are shown as normalised potential impacts for a short-term (0 – 100 years) time horizon

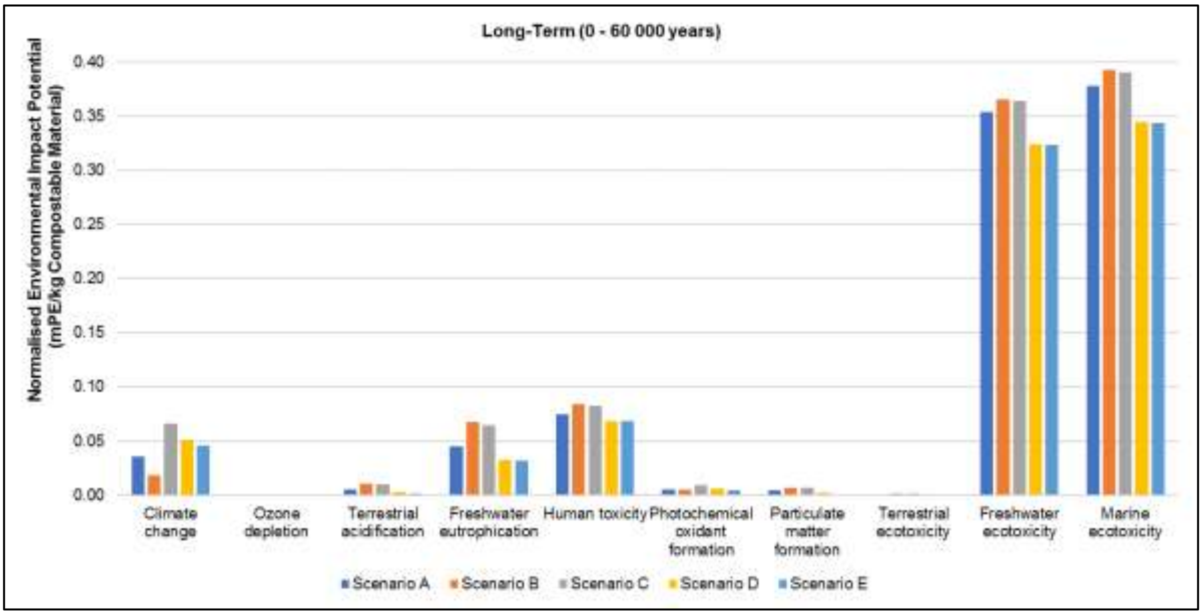

Figure A.5 LCIA results for the treatment of 1kg of compostable organic material in the five landfill scenarios investigated. Results are shown as normalised potential impacts for a long-term (0 – 60 000 years) time horizon

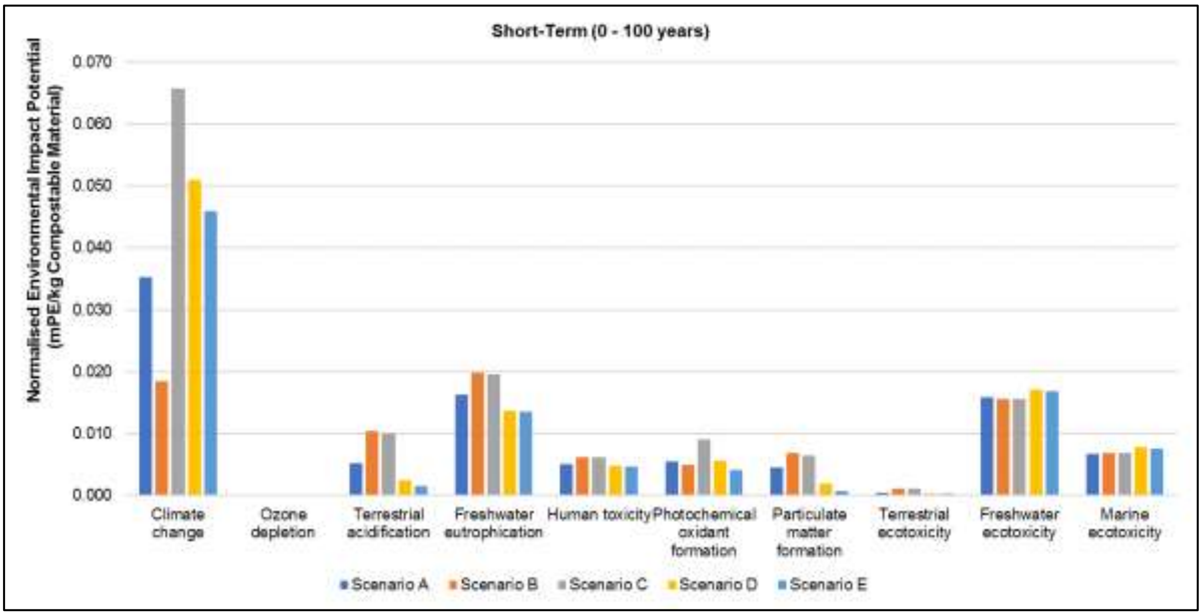

Figure A.6 LCIA results for the treatment of 1kg of compostable organic material in the five landfill scenarios investigated. Results are shown as normalised potential impacts for a short-term (0 – 100 years) time horizon

Table A.12 Quantified LCIA results for the treatment of 1kg of polyethylene, cardboard and compostable material in the five landfill scenarios investigated. Results are shown as normalised potential impacts

	Polyethylene					**Cardboard**					**Compostable Material**				
Scenario (mPE/kg)	**A**	**B**	**C**	**D**	**E**	**A**	**B**	**C**	**D**	**E**	**A**	**B**	**C**	**D**	**E**
Climate change	9.68E-03	6.57E-03	1.53E-02	1.24E-02	9.70E-03	1.08E-01	5.30E-02	2.05E-01	1.62E-01	1.21E-01	3.53E-02	1.85E-02	6.57E-02	5.09E-02	4.59E-02
Ozone depletion	1.21E-04	1.22E-04	1.21E-04	3.58E-05	0.00E+00	1.58E-04	1.28E-04	1.28E-04	3.58E-05	0.00E+00	1.62E-04	1.32E-04	1.31E-04	3.58E-05	0.00E+00
Terrestrial acidification	1.66E-03	2.47E-03	2.08E-03	1.03E-03	1.62E-05	4.39E-03	9.89E-03	9.49E-03	3.20E-03	2.19E-03	5.20E-03	1.05E-02	1.01E-02	2.55E-03	1.54E-03
Freshwater eutrophication	2.09E-03	5.95E-03	3.43E-03	5.06E-04	0.00E+00	4.74E-03	2.89E-02	2.64E-02	5.06E-04	0.00E+00	4.48E-02	6.73E-02	6.48E-02	3.24E-02	3.19E-02
Human toxicity	6.59E-01	6.65E-01	6.64E-01	6.59E-01	6.58E-01	2.11E-01	2.01E-01	2.00E-01	2.12E-01	2.11E-01	7.46E-02	8.38E-02	8.27E-02	6.85E-02	6.81E-02
Photochemical oxidant formation	2.07E-03	2.07E-03	2.74E-03	1.89E-03	6.36E-04	1.16E-02	7.89E-03	2.13E-02	1.56E-02	1.10E-02	5.52E-03	5.02E-03	9.10E-03	5.64E-03	4.10E-03
Particulate matter formation	1.91E-03	2.53E-03	2.25E-03	1.21E-03	7.48E-06	3.77E-03	7.04E-03	6.08E-03	2.21E-03	1.01E-03	4.49E-03	6.90E-03	6.43E-03	1.91E-03	7.09E-04
Terrestrial ecotoxicity	7.63E-04	2.32E-03	2.31E-03	7.06E-04	6.87E-04	1.45E-04	2.52E-03	2.52E-03	6.86E-05	4.97E-05	6.06E-04	1.23E-03	1.23E-03	4.57E-04	4.39E-04
Freshwater ecotoxicity	6.40E+00	6.40E+00	6.40E+00	6.41E+00	6.41E+00	6.28E-01	6.44E-01	6.42E-01	6.22E-01	6.21E-01	3.54E-01	3.66E-01	3.64E-01	3.24E-01	3.24E-01
Marine ecotoxicity	7.87E+00	7.87E+00	7.86E+00	7.87E+00	7.87E+00	7.12E-01	7.29E-01	7.27E-01	7.05E-01	7.04E-01	3.78E-01	3.93E-01	3.90E-01	3.45E-01	3.44E-01

Table A.13 Quantified LCIA results for the treatment of 1kg of polyethylene, cardboard and compostable material in the five landfill scenarios investigated. Results are shown as normalised potential impacts for a short-term (0 – 100 years) time horizon

Screened List	**Polyethylene**					**Cardboard**					**Organic/Compostable Material**				
Scenario (mPE/kg)	**A**	**B**	**C**	**D**	**E**	**A**	**B**	**C**	**D**	**E**	**A**	**B**	**C**	**D**	**E**
Climate change	9.68E-03	6.57E-03	1.53E-02	1.24E-02	9.70E-03	1.08E-01	5.30E-02	2.05E-01	1.62E-01	1.21E-01	3.53E-02	1.85E-02	6.57E-02	5.09E-02	4.59E-02
Ozone depletion	1.21E-04	1.22E-04	1.21E-04	3.58E-05	0.00E+00	1.58E-04	1.28E-04	1.28E-04	3.58E-05	0.00E+00	1.62E-04	1.32E-04	1.31E-04	3.58E-05	0.00E+00
Terrestrial acidification	1.66E-03	2.47E-03	2.08E-03	1.03E-03	1.62E-05	4.39E-03	9.89E-03	9.49E-03	3.20E-03	2.19E-03	5.20E-03	1.05E-02	1.01E-02	2.55E-03	1.54E-03
Freshwater eutrophication	3.78E-04	8.80E-04	5.60E-04	8.68E-05	0.00E+00	8.25E-04	3.90E-03	3.58E-03	8.68E-05	0.00E+00	1.63E-02	1.99E-02	1.95E-02	1.37E-02	1.36E-02
Human toxicity	1.88E-03	7.27E-03	7.20E-03	3.88E-03	3.74E-03	2.50E-02	3.46E-02	3.46E-02	5.76E-02	5.74E-02	5.04E-03	6.19E-03	6.13E-03	4.83E-03	4.68E-03
Photochemical oxidant formation	2.07E-03	2.07E-03	2.74E-03	1.89E-03	6.36E-04	1.16E-02	7.89E-03	2.13E-02	1.56E-02	1.10E-02	5.52E-03	5.02E-03	9.10E-03	5.64E-03	4.10E-03
Particulate matter formation	1.91E-03	2.53E-03	2.25E-03	1.21E-03	7.48E-06	3.76E-03	7.04E-03	6.07E-03	2.21E-03	1.01E-03	4.49E-03	6.90E-03	6.43E-03	1.91E-03	7.09E-04
Terrestrial ecotoxicity	9.44E-05	1.65E-03	1.65E-03	3.72E-05	1.82E-05	1.20E-04	2.50E-03	2.50E-03	4.59E-05	2.70E-05	4.67E-04	1.09E-03	1.09E-03	3.20E-04	3.01E-04
Freshwater ecotoxicity	1.18E-02	1.23E-02	1.23E-02	2.22E-02	2.20E-02	1.09E-02	1.08E-02	1.08E-02	2.48E-02	2.46E-02	1.58E-02	1.56E-02	1.56E-02	1.71E-02	1.69E-02
Marine ecotoxicity	1.42E-02	1.48E-02	1.48E-02	2.70E-02	2.68E-02	1.37E-02	1.36E-02	1.35E-02	3.04E-02	3.02E-02	6.74E-03	6.89E-03	6.85E-03	7.78E-03	7.54E-03

Table A.14 Percentage contribution of short-term potential impacts (0 -100 years) towards overall potential impacts (0 – 60 000 years) for various standard and toxicity related environmental impact categories

Polyethylene					**Cardboard**					**Compostable Material**				
Landfill Scenario					**Landfill Scenario**					**Landfill Scenario**				
A	**B**	**C**	**D**	**E**	**A**	**B**	**C**	**D**	**E**	**A**	**B**	**C**	**D**	**E**
% Contribution of short-term impacts														
Climate change														
100	100	100	100	100	100	100	100	100	100	100	100	100	100	100
Ozone depletion														
100	100	100	100	n/a	100	100	100	100	n/a	100	100	100	100	n/a
Terrestrial acidification														
100	100	100	100	100	100	100	100	100	100	100	100	100	100	100
Freshwater eutrophication														
18	15	16	17	n/a	17	14	14	17	n/a	36	30	30	42	43
Marine eutrophication														
2	2	2	2	2	77	77	77	79	79	62	62	62	65	65
Human toxicity														
0	1	1	1	1	12	17	17	27	27	7	7	7	7	7
Photochemical oxidant formation														
100	100	100	100	100	100	100	100	100	100	100	100	100	100	100
Particulate matter formation														
100	100	100	100	100	100	100	100	100	100	100	100	100	100	100
Terrestrial ecotoxicity														
12	71	71	5	3	83	99	99	67	54	77	89	89	70	69
Freshwater ecotoxicity														
0	0	0	0	0	2	2	2	4	4	4	4	4	5	5
Marine ecotoxicity														
0	0	0	0	0	2	2	2	4	4	2	2	2	2	2

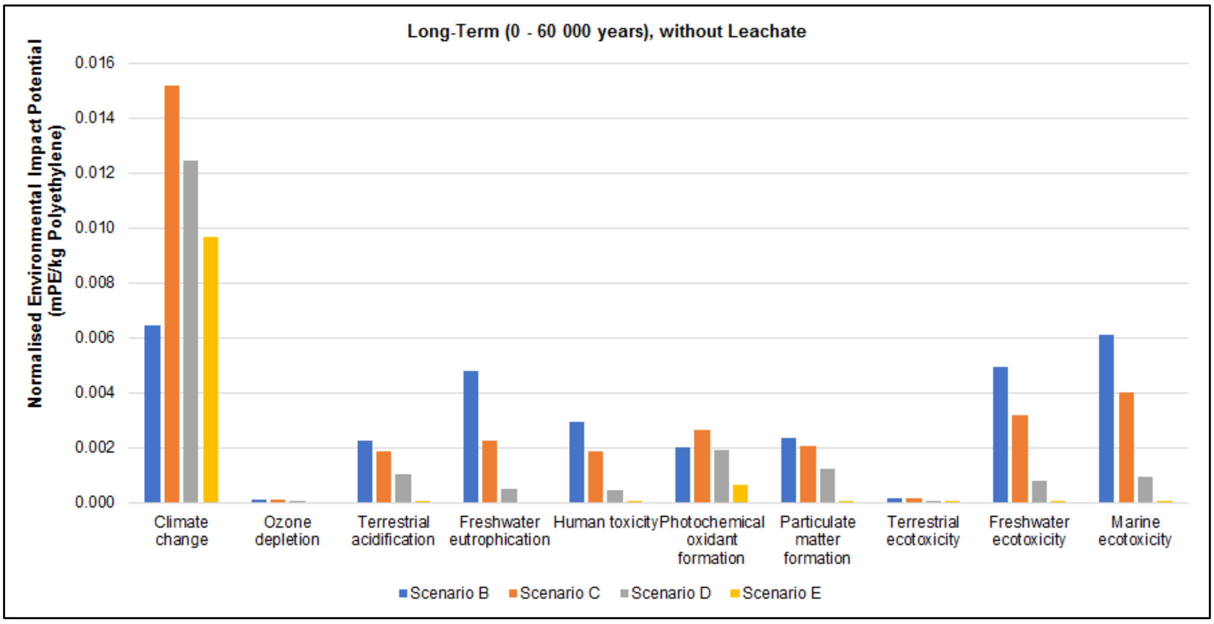

Figure A.7 LCIA results for the treatment of 1kg of polyethylene in the four South African specific landfill scenarios investigated with and without leachate generation and release. Results are shown as normalised potential impacts for the long-term (0 – 60 000 years) time horizon

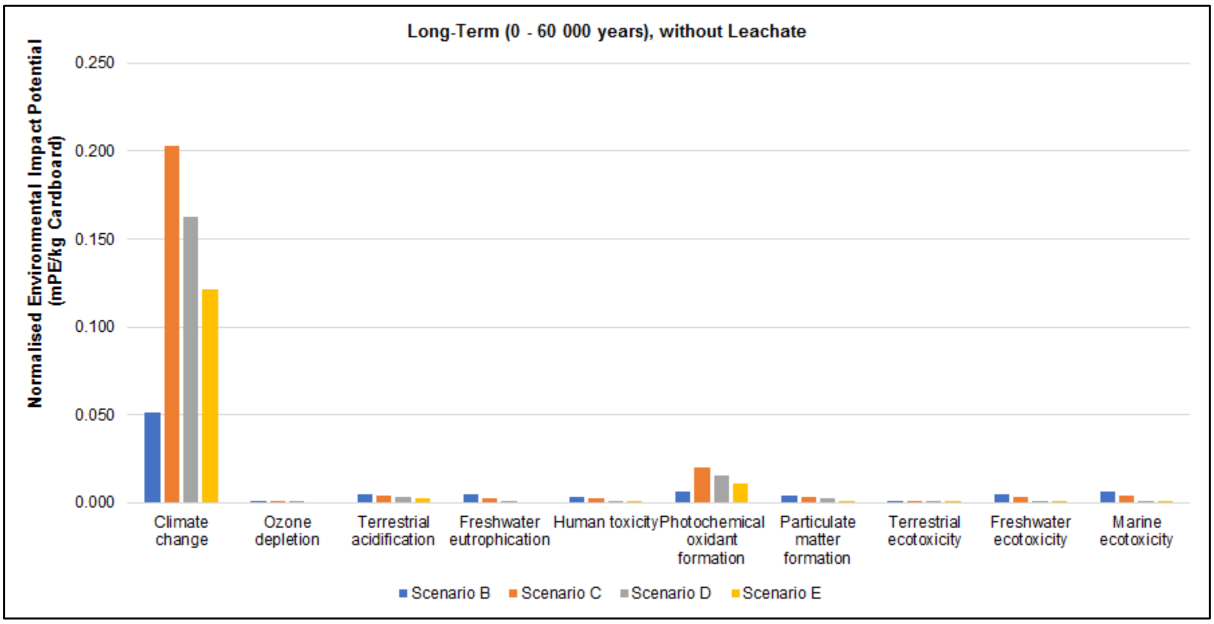

Figure A.8 LCIA results for the treatment of 1kg of cardboard in the four South African specific landfill scenarios investigated with and without leachate generation and release. Results are shown as normalised potential impacts for the long-term (0 – 60 000 years) time horizon

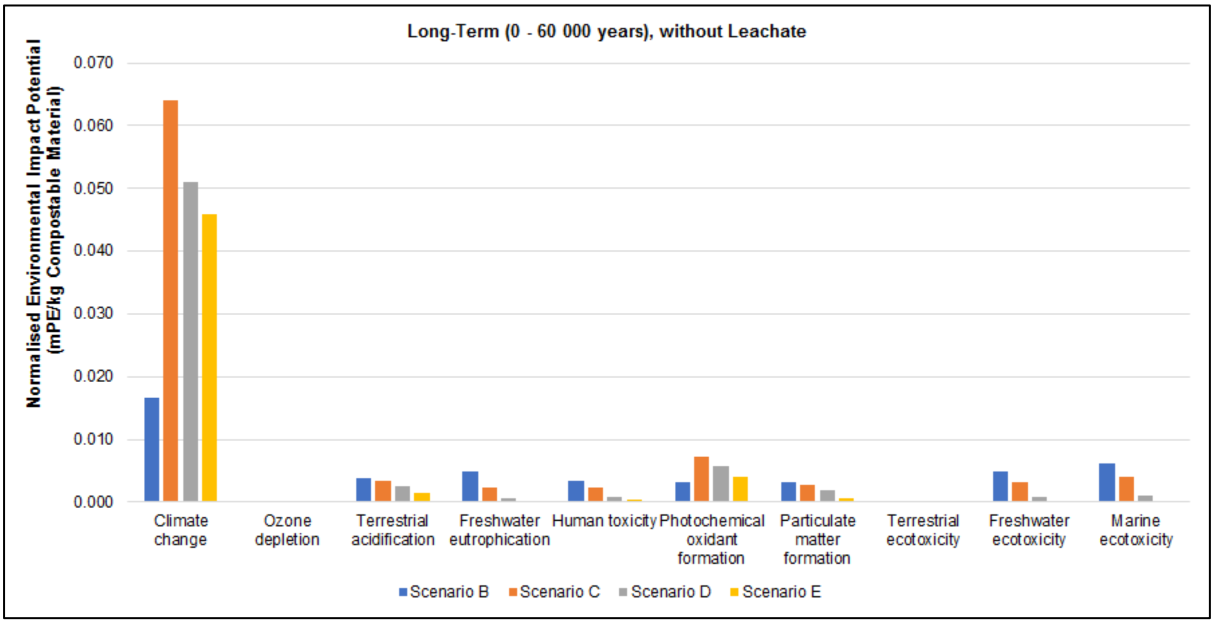

Figure A.9 LCIA results for the treatment of 1kg of compostable material in the four South African specific landfill scenarios investigated with and without leachate generation and release. Results are shown as normalised potential impacts for the long-term (0 – 60 000 years) time horizon

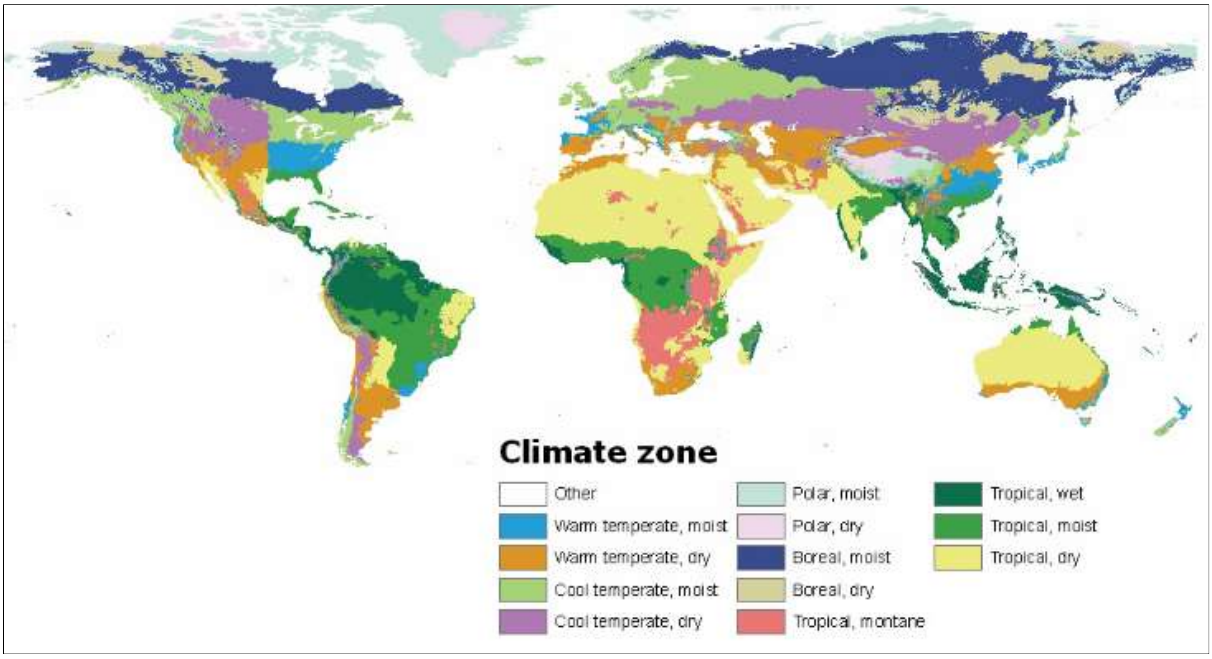

Figure A.10 Climate zones according to IPCC classification (European Commission - Joint Research Centre Institute for Environment and Sustainability, 2014)

APPENDIX B

SUPPLEMENTARY LITERATURE

B.1 Determination of Landfill GHG Emissions

The determination of an emission factor for the GHG emissions arising from the waste body has been presented in detail by both the IPCC (2006b) and Manfredi et al. (2009a). While these approaches are largely consistent, the dissimilation factors used by Manfredi et al. (2009a) provide an additional level of detail to the calculation output when compared to the DOC_f defined by the IPCC (2006b). For the 100 year time horizon considered in this approach, it is assumed that the amount of biogenic carbon contained in the waste is gradually reduced due to gas and leachate emissions. It is further assumed that on a mass basis, 55% of the emitted carbon is released as CH_4 with the remainder emitted as CO_2 (Manfredi et al., 2009a). Thus, the total amount of CH_4 and CO_2 generated from the waste body (in kg tonne^{-1} wet waste) can be determined from Equation B.1 and Equation B.2 (adapted from Manfredi et al. (2009a:829)) as follows:

$$CH_{4_{generated}} = DOC \times D_{LFG} \times \frac{55}{100} \times \frac{16}{12}$$

Equation B.1

$$CO_{2_{generated}} = CH_{4_{generated}} \times \frac{45}{55}$$

Equation B.2

For landfills where generated LFG is captured and treated (typically either through flaring or utilized for energy generation), additional parameters are defined to account for various efficiencies in collection and utilisation. These are discussed in detail in both Manfredi et al. (2009a) and IPCC (2006b).

GHG emissions from the waste body are consequent of the carbon content of the waste itself. While Manfredi et al. (2009a) favour the term *biogenic carbon content,* conventionally, this parameter is referred to as *degradable organic carbon* (DOC) (i.e. IPCC (2006a)). Within the landfill, the biogenic carbon, or DOC, is degraded and emitted via gas or leachate, typically accounted for with the use of dissimilation coefficients (D_{LFG} and $D_{Leachate}$) (Manfredi et al., 2009a:829). The IPCC approach assumes that the carbon lost in leachate is negligible, thus a degradability factor (DOC_f) is used to represent the degraded carbon, defined as the fraction of carbon degraded and released as LFG from the waste (IPCC, 2006b:3.13). Various studies have been undertaken to characterise these parameters for different waste types. A summary of these studies is available in Table B.1 (Appendix B, Section B.2).

Whilst the biogenic carbon content of the waste is an inherent property of the waste type, the dissimilation factor is dependent on both the waste type and landfill conditions. It has been suggested that water availability on a landfill often “constitutes the limiting factor for waste degradation”, resulting in reduced emissions of both LFG and leachate from the waste body (Manfredi et al., 2009a:829). Therefore, degradability factors derived for conditions with high water availability might not be applicable to conditions where water supply is limited. Even in cases where water supply is not considered deficient, some studies consider the carbon losses in the landfill leachate so small as to be insignificant in GHG accounting (e.g. IPCC (2006b)). Manfredi et al. (2009a:827) refute this assumption, suggesting that leachate can contain 1 – 4% of the biogenic carbon contained in the waste. While the leached carbon has no direct global warming potential, neglecting this loss can result in the overestimation of carbon sequestered in the landfill (Manfredi et al., 2009a).

B.2 Biogenic Carbon Content and Dissimilation Factors for Different Waste Types

A summary of the results of studies to characterise the biogenic carbon content and dissimilation factor for different materials contained in waste is shown in Table B.1 (as reported by Manfredi et al. (2009a)).

Comparison of the biogenic carbon content for different waste types in Table B.1 to corresponding categories presented as default DOC values by the IPCC (2006a:2.14) (2006a), shows that although figures vary slightly, they typically fall within the IPCC range. While DOC is recognised as one of the main parameters affecting CH_4 emissions from landfill, limitations in both the availability of studies investigating this parameter and inconsistencies in the sampling methodologies used introduce a level of uncertainty into the reported values (IPCC, 2006a). For the IPCC default values, uncertainty is reported at approximately 20% while for country-specific values based on representative sampling and analysis, this uncertainty is reduced to approximately 10% (IPCC, 2006b:3.27)

Table B.1 Typical ranges of biogenic carbon contents and dissimilation factors of various waste fractions (As presented in Manfredi et al., 2009a:828)[1]

Material fraction	Biogenic carbon content (kg carbon.tonne^{-1} wet waste)	Dissimilation factor of biogenic carbon as LFG (D_{LFG})
Household waste (all fractions)	160 – 200	0.50
Kitchen organics	100 – 120	0.64
Newspapers	360 – 440	0.2
Office paper	300 – 360	0.88
Cardboard	300 – 380	0.45
Wood	400 – 450	0.23
Plastic	0 (650 – 750 of carbon fossil)	0
Glass	0	0
Metals	0	0
Predominantly mineral waste	12 – 25	0

[1]Summarised results of the studies undertaken by Barlaz (1998), Eleazer et al. (1997), Manfredi et al. (2009b), Riber, Petersen & Christensen (2009) and EPA (2006) as presented in Manfredi et al. (2009a).

B.3 Use of a Methane Correction Factor in Determining Greenhouse Gas Emissions

While the use of the MCF is recommended by IPCC methodology, two major sources of uncertainty are associated with the use of this factor (IPCC, 2006b:3.26).

1. The value of the MCF for each type of site is based on “one experimental study and expert judgement” as opposed to measured data.
2. Site classification is based on “expert opinion” and consequently it is unlikely that any countries will be able to classify their unmanaged sites into the appropriate categories based on available data. Furthermore, uncertainty in site classification might challenge countries in identifying sites that meet the IPCC criteria for managed sites.

Of additional concern is the definition of an appropriate MCF to use for developing countries in tropical and sub-tropical climate zones (Frøiland Jensen & Pippatti, 2001). Based on the findings from numerous site observations from unmanaged or poorly managed disposal sites in developing countries in these climate zones, the organic fraction of disposed material is typically fully degraded two to seven years after disposal (Frøiland Jensen & Pippatti, 2001:426). Consequently, the default value of 0.6 for unmanaged sites could be too high under these conditions. Site observations of unmanaged or poorly managed landfills further suggest that burning is a widespread practice in many developing countries (Frøiland Jensen & Pippatti, 2001). Uncontrolled burning can be attributed to accidental fires or fires started by scavengers, but can also be an intentional management strategy to reduce waste volumes and address other concerns such as vermin and odours (Frøiland Jensen & Pippatti, 2001). The effect of such fires is seen in the reduction of the CH_4 emission potential of the site, suggesting that the MCF for the site should be lower than those defined in Table 2.9 (Frøiland Jensen & Pippatti, 2001).

APPENDIX C

SUPPLEMENTARY METHODOLOGY

C.1 Supplementary Information for Stage 1

Table C.1 Government legislation, regulations and standards reviewed during Stage 1 of the research methodology

Legislation
National Environmental Management: Waste Act, No. 59 of 2008 (2009)
National Environmental Management: Waste Act, No. 59 of 2008. National Waste Management Strategy (2012)
National Environmental Management: Waste Amendment Act, No. 26 of 2014 (2014)
National Environmental Management: Waste Act, No. 59 of 2008. National Pricing Strategy for Waste Management (2016)
National Environmental Management: Waste Act, No. 59 of 2008. National Domestic Waste Collection Standards (2011)
National Environmental Management: Waste Act, No. 59 of 2008 National Norms and Standards for Disposal of Waste to Landfill (2013)
National Environmental Management: Waste Act, No. 59 of 2008. National Waste Information Regulations (2012)
National Environmental Management: Waste Act, No. 59 of 2008. Waste Classification and Management Regulations (2013)
National Environmental Management: Waste Act, No. 59 of 2008. List of Waste Management Activities that have, or are likely to have, a Detrimental Effect on the Environment (2013)
National Environmental Management: Waste Act, No. 59 of 2008. National Norms and Standards for the Assessment of Waste for Landfill Disposal (2013)
Municipal Waste Sector Plan (DEA, 2011b)
Minimum Requirements for the Handling, Classification and Disposal of Hazardous Waste (DWAF, 1998a)
Minimum Requirements for Waste Disposal by Landfill (DWAF, 1998c)
Minimum Requirements for Water Monitoring at Waste Facilities (DWAF, 1998c*)*

Table C.2 Waste management facilities visited during the field work phase of the research

Site	Description
Coastal Park Landfill Site, Muizenberg, Cape Town	G:L:B landfill site. No wastewater treatment works on site and no landfill gas capture
Simmer and Jack Landfill, Germiston	G:L:B landfill site. No wastewater treatment works on site but 1MW gas to electricity plant operating on site
Athlone Transfer Station	Waste transfer station accepting general waste and low volumes of hazardous waste. Recyclables and green waste separated from waste stream
Kraaifontein Waste Management Facility	Integrated waste management facility and transfer station accepting general waste and low volumes of hazardous waste. Recyclables and green waste separated from waste stream
Waste to Want plastic recovery facility, Elsies River	PET plastic bottle collection and recycling facility
Oasis Association, Cape Town	Recycling collection and sorting facility
Actonville Recycling Facility, Actonville	Recycling collection and sorting facility
Uilenkraal Dairy Farm, Darling, Biogas plant	Biogas generation from dairy farm organic material
Cape Dairy Biogas Project, Malmesbury	Biogas generation from dairy farm organic material

Table C.3 Overview of stakeholders consulted during the field work component of the research

Interviewee	Position and Organisation	Main Points of Discussion
Sally-Anne Käsner	Executive Associate (Environmental Management and Sustainability) at JG Afrika	Waste management facility compliance and auditing
Eddie Hanekom	Director, Waste Management, Western Cape Government: Department of Environmental Affairs and Waste Management	National and provincial waste management standards and reporting
Adiel De Bruyn	Solid Waste Management Department, Disposal Branch, City of Cape Town	Landfill site operations and standards
Owen Jacobs	Facility Manager, Athlone Waste Transfer Station	Facility operations and standards within the context of the South African waste sector
Arlene Van Staden	City of Cape Town	South African waste sector
Margot Ladouce	Head, Research and Development, Monitoring and Auditing. Solid Waste Management Department, Disposal Branch, City of Cape Town	Provincial and municipal waste management standards and reporting

C.2 Parameterisation of Ecoinvent Sanitary Landfill Model

Parameterisation of waste composition

1. Open sheet 13_MSWIv2.xls
2. Open sheet 13_MSWLFv2.xls
3. Select sheet “waste input” on 13_MSWIv2.xls
4. Specify intended material fraction in waste in row 12 (fraction of total waste mixture)
5. Specify burnable (1) or inert (0) in row 13

Parameterisation of LFG capture, flaring and utilisation efficiencies

1. Open sheet 13_MSWIv2.xls
2. Open sheet 13_MSWLFv2.xls
3. Select sheet “air and energy” on 13_MSWLFv2.xls
4. Specify LFG capture efficiency (as a percentage) in cell D7
5. Specify utilisation efficiency (as a percentage) in cell F7

C.3 Modelling Parameters

Table C.4 Process-specific burdens used for different landfill scenarios modelled

	CH Sanitary Landfill	**South African Sanitary Landfill**		**South African Unsanitary Landfill**	
	Ecoinvent default conditions	**LFG Capture and Flaring**	**Direct LFG Emissions**	**Covered, unsanitary**	**Uncovered, open dump**
	A	**B**	**C**	**D**	**E**
CH Sanitary landfill dataset	Sanitary landfill facility (CH) | construction | Alloc Rec, U				
South African/proxy dataset	Sanitary landfill facility (GLO) | market for | Alloc Rec, U				
Inventory Demand	5.56E-10	5.56E-10	5.56E-10	n/a	n/a
CH Sanitary landfill dataset	Process-specific burden, sanitary landfill (CH) | market for process-specific burden, sanitary landfill | Alloc Rec, U				
South African/proxy dataset	Process-specific burden, sanitary landfill (RoW) | processing, sanitary landfill | Alloc Rec, U				
Inventory Demand	1	1	1	1	1[a]
CH Sanitary landfill dataset	Electricity, low voltage (CH)| market for | Alloc Rec, U[b]				
South African/proxy dataset	Electricity, low voltage (ZA)| market for | Alloc Rec, U[b]				
Inventory Demand	0.00864 kWh	0.00864 kWh	0.00864 kWh	n/a	n/a
CH Sanitary landfill dataset	Heat, central or small-scale, other than natural gas (CH) | market for | Alloc Rec, U[c]				
South African/proxy dataset	Heat, central or small-scale, other than natural gas (RoW) | market for | Alloc Rec, U[c]				
Inventory Demand	0.00054238 MJ	0.00054238 MJ	0.00054238 MJ	n/a	n/a

Continuation of Table C.4 Process-specific burdens used for different landfill scenarios modelled

	CH Sanitary Landfill	South African Sanitary Landfill		South African Unsanitary Landfill	
	Ecoinvent default conditions	LFG Capture and Flaring	Direct LFG Emissions	Covered, unsanitary	Uncovered, open dump
	A	B	C	D	E
CH Sanitary landfill dataset	Wastewater treatment facility, capacity 5E9l/year (CH)| market for wastewater treatment facility, capacity 5E9l/year | Alloc Rec, U				
South African/proxy dataset	Wastewater treatment facility, capacity 5E9l/year (RoW)| market for wastewater treatment facility, capacity 5E9l/year | Alloc Rec, U				
Inventory Demand	1.42E-11 p	1.42E-11 p	1.42E-11 p	n/a	n/a
CH Sanitary landfill dataset	Iron sulfate (GLO)| market for | Alloc Rec, U				
South African/proxy dataset	Iron sulfate (GLO)| market for | Alloc Rec, U				
Inventory Demand	6.43E-5 kg	6.43E-5 kg	6.43E-5 kg	n/a	n/a
CH Sanitary landfill dataset	Sewer grid, 5E9l/year, 110 km (CH)| market for sewer grid, 5E9l/year, 110 km | Alloc Rec, U				
South African/proxy dataset	Sewer grid, 5E9l/year, 110 km (RoW)| market for sewer grid, 5E9l/year, 110 km | Alloc Rec, U				
Inventory Demand	5.45E-10 km	5.45E-10 km	5.45E-10 km	n/a	n/a
CH Sanitary landfill dataset	Aluminium sulfate, powder (GLO)| market for | Alloc Rec, U				
South African/proxy dataset	Aluminium sulfate, powder (GLO)| market for | Alloc Rec, U				
Inventory Demand	1.74E-5 kg	1.74E-5 kg	1.74E-5 kg	n/a	n/a

[a] Modified dataset containing only ‘Known inputs from nature (resources)’ as specified in the Process-specific burden, sanitary landfill (CH) | market for process-specific burden, sanitary landfill | Alloc Rec, U dataset

[b] Electricity demand for wastewater treatment of short-term leachate

[c] Heat for heating the digester and general space heating

Table C.5 Process-specific burdens contained in the Process-specific burden, sanitary landfill (CH) | market for process-specific burden, sanitary landfill | Alloc Rec, U dataset (excluding all known inputs from nature (resources))

	CH Sanitary Landfill	**South African Sanitary Landfill**		**South African Unsanitary Landfill**	
	Ecoinvent default conditions	**LFG Capture and Flaring**	**Direct LFG Emissions**	**Covered, unsanitary**	**Uncovered, open dump**
	A	**B**	**C**	**D**	**E**
CH Sanitary landfill dataset	Diesel, burned in building machine (GLO)| market for | Alloc Rec, U[a]				
South African/proxy dataset	Diesel, burned in building machine (GLO)| market for | Alloc Rec, U[a]				
Inventory Demand	*0.0467 MJ (1.3 litres/tonne waste)*	*0.0467 MJ (1.3 litres/tonne waste)*	*0.0467 MJ (1.3 litres/tonne waste)*	*0.0467 MJ (1.3 litres/tonne waste)*	n/a
CH Sanitary landfill dataset	Heat, central or small-scale, other than natural gas (CH)| market group for | Alloc Rec, U[b]				
South African/proxy dataset	*n/a*				
Inventory Demand	*0.0015134 MJ*	n/a	n/a	n/a	n/a
CH Sanitary landfill dataset	Electricity, medium voltage (CH)| market for | Alloc Rec, U[c]				
South African/proxy dataset	Electricity, medium (ZA)| market for | Alloc Rec, U[c]				
Inventory Demand	*0.00135 kWh*	*0.00135 kWh*	n/a	n/a	n/a
CH Sanitary landfill dataset	Electricity, low voltage (CH)| market for | Alloc Rec, U[d]				
South African/proxy dataset	Electricity, low (ZA)| market for | Alloc Rec, U[d]				
Inventory Demand	*1.5E-5 kWh*	*1.5E-5 kWh*	*1.5E-5 kWh*	*1.5E-5 kWh*	n/a

[a] Diesel consumption of special loaders used to distribute and compact the waste.

[b] Energy demand of simple administrative house

[c] Operation of landfill gas pumps

[d] Energy demand of simple administrative houses

Table C.6 Inventory values for polyethylene, cardboard and compostable material waste as specified in the ecoinvent incineration model

	Unit	Mean Value	Uncertainty GSD	Contribution to total from Burnable Fraction	Mean Value	Uncertainty GSD	Contribution from Burnable Fraction	Mean Value	Uncertainty GSD	Contribution from Burnable Fraction
		Polyethylene: Degradability = 1%			**Cardboard: Degradability = 32.4%**			**Compostable Material (Degradability = 27%)**		
Upper heating value	MJ/kg	42.82	113%		17.91	123%		5.72	135%	
Lower heating value	MJ/kg	42.47	113%		15.92	125%		4.00	100%	
Water content	kg/kg waste	0.004	200%		0.104	141%		0.6	109%	
Oxygen (without O from H2O)	kg/kg waste	3.84E-02	159%		3.93E-01	117%		1.26E-01	137%	
Hydrogen (without H from H2O)	kg/kg waste	1.22E-01	138%		5.78E-02	152%		2.00E-02	171%	
Carbon	kg/kg waste	8.22E-01	104%	8.22E-01	4.33E-01	115%	4.33E-01	1.62E-01	133%	1.62E-01
Sulfur	kg/kg waste	4.27E-04	240%	4.27E-04	1.78E-03	215%	1.78E-03	1.50E-03	218%	1.50E-03
Nitrogen	kg/kg waste	1.30E-03	220%	1.30E-03	2.58E-03	208%	2.58E-03	4.00E-03	200%	4.00E-03
Phosphor	kg/kg waste	0	100%	0	0	100%	0	1.13E-03	223%	1.13E-03
Boron	kg/kg waste	0	100%	0	0	100%	0	1.02E-05	308%	1.02E-05
Chlorine	kg/kg waste	1.46E-03	218%	1.46E-03	7.17E-03	189%	7.17E-03	4.00E-03	200%	4.00E-03
Bromium	kg/kg waste	9.59E-06	309%	9.59E-06	0	100%	0	6.00E-06	318%	6.00E-06
Fluorine	kg/kg waste	1.44E-05	302%	1.44E-05	0	100%	0	2.00E-04	254%	2.00E-04
Iodine	kg/kg waste	0	100%	0	0	100%	0	5.50E-08	400%	5.50E-08

Continuation of Table C.6 Inventory values for polyethylene, cardboard and compostable material waste as specified in the ecoinvent incineration model

	Unit	Mean Value	Uncertainty GSD	Contribution from Burnable Fraction	Mean Value	Uncertainty GSD	Contribution from Burnable Fraction	Mean Value	Uncertainty GSD	Contribution from Burnable Fraction
		Polyethylene			**Cardboard**			**Compostable Material**		
Silver	kg/kg waste	0	100%	0	0	100%	0	0.00E+00	100%	0
Arsenic	kg/kg waste	1.82E-06	339%	1.82E-06	1.2E-06	347%	1.17E-06	2.00E-06	337%	2.00E-06
Barium	kg/kg waste	2.40E-04	251%	2.40E-04	5.7E-05	277%	5.72E-05	0.00E+00	100%	0
Cadmium	kg/kg waste	3.39E-05	286%	3.39E-05	9.3E-07	351%	9.33E-07	1.38E-07	386%	1.38E-07
Cobalt	kg/kg waste	1.68E-06	341%	1.68E-06	9.2E-07	352%	9.18E-07	5.00E-06	321%	5.00E-06
Chromium	kg/kg waste	1.25E-05	304%	1.25E-05	9.2E-06	310%	9.24E-06	8.00E-06	312%	8.00E-06
Copper	kg/kg waste	4.14E-05	283%	4.14E-05	3.5E-05	286%	3.51E-05	1.80E-05	298%	1.80E-05
Mercury	kg/kg waste	4.79E-08	400%	4.79E-08	3E-07	372%	3.01E-07	7.00E-08	400%	7.00E-08
Manganese	kg/kg waste	3.00E-05	288%	3.00E-05	9.5E-05	268%	9.52E-05	4.30E-06	324%	4.30E-06
Molybdenum	kg/kg waste	0	100%	0	0	100%	0	4.00E-07	367%	4.00E-07
Nickel	kg/kg waste	9.59E-07	351%	9.59E-07	1.1E-05	307%	1.06E-05	5.42E-06	319%	5.42E-06
Lead	kg/kg waste	2.26E-05	294%	2.26E-05	2.9E-05	289%	2.90E-05	1.86E-05	297%	1.86E-05
Antimony	kg/kg waste	1.01E-05	308%	1.01E-05	4.6E-09	400%	4.60E-09	0.00E+00	100%	0
Selenium	kg/kg waste	2.01E-06	337%	2.01E-06	3.1E-06	329%	3.11E-06	5.00E-07	363%	5.00E-07
Tin	kg/kg waste	3.81E-06	326%	3.81E-06	6.8E-06	315%	6.75E-06	8.00E-06	312%	8.00E-06
Vanadium	kg/kg waste	2.21E-03	211%	2.21E-03	0	100%	0	3.00E-06	330%	3.00E-06
Zinc	kg/kg waste	3.42E-04	244%	3.42E-04	6.3E-05	275%	6.33E-05	5.82E-05	276%	5.82E-05
Beryllium	kg/kg waste	4.79E-07	363%	4.79E-07	1.1E-06	348%	1.13E-06	0	100%	0
Scandium	kg/kg waste	0	100%	0	0	100%	0	0	100%	0
Strontium	kg/kg waste	8.49E-05	270%	8.49E-05	4.5E-05	281%	4.50E-05	0	100%	0
Titanium	kg/kg waste	9.59E-04	226%	9.59E-04	0	100%	0	0	100%	0

Continuation of Table C.6 Inventory values for polyethylene, cardboard and compostable material waste as specified in the ecoinvent incineration model

	Unit	Mean Value	Uncertainty GSD	Contribution from Burnable Fraction	Mean Value	Uncertainty GSD	Contribution from Burnable Fraction	Mean Value	Uncertainty GSD	Contribution from Burnable Fraction
		Polyethylene			**Cardboard**			**Compostable Material**		
Scandium	kg/kg waste	0	100%	0	0	100%	0	0	100%	0
Strontium	kg/kg waste	8.49E-05	270%	8.49E-05	4.5E-05	281%	4.50E-05	0	100%	0
Titanium	kg/kg waste	9.59E-04	226%	9.59E-04	0	100%	0	0	100%	0
Thallium	kg/kg waste	3.84E-07	367%	3.84E-07	0	100%	0	0	100%	0
Tungsten	kg/kg waste	0	100%	0	0	100%	0	0	100%	0
Silicon	kg/kg waste	0	100%	0	0	100%	0	4.00E-02	158%	4.00E-02
Iron	kg/kg waste	1.53E-03	217%	1.53E-03	0	100%	0	6.00E-04	234%	6.00E-04
Calcium	kg/kg waste	2.59E-03	208%	2.59E-03	0	100%	0	2.18E-02	169%	2.18E-02
Aluminium	kg/kg waste	1.92E-04	255%	1.92E-04	0	100%	0	1.00E-02	183%	1.00E-02
Potassium	kg/kg waste	0	100%	0	0	100%	0	3.50E-03	202%	3.50E-03
Magnesium	kg/kg waste	9.59E-05	267%	9.59E-05	0	100%	0	2.82E-03	206%	2.82E-03
Sodium	kg/kg waste	1.41E-03	219%	1.41E-03	0	100%	0	1.50E-03	218%	1.50E-03

APPENDIX D

Application for Approval of Ethics in Research (EiR) Projects
Faculty of Engineering and the Built Environment, University of Cape Town

APPLICATION FORM

Please Note:
Any person planning to undertake research in the Faculty of Engineering and the Built Environment (EBE) at the University of Cape Town is required to complete this form **before** collecting or analysing data.The objective of submitting this application *prior* to embarking on research is to ensure that the highest ethical standards in research, conducted under the auspices of the EBE Faculty, are met. Please ensure that you have read, and understood the **EBE Ethics in Research Handbook**(available from the UCT EBE, Research Ethics website) prior to completing this application form: http://www.ebe.uct.ac.za/usr/ebe/research/ethics.pdf

APPLICANT'S DETAILS		
Name of principal researcher, student or external applicant		Clare Rodseth
Department		Chemical engineering
Preferred email address of applicant:		clare.rodseth@gmail.com
If a Student	Your Degree: e.g., MSc, PhD, etc.,	MSc. Eng
	Name of Supervisor (if supervised):	Prof. Harro Von Blottnitz
If this is a researchcontract, indicate the source of funding/sponsorship		
Project Title		Methods for Product Life-Cycle Management in African Markets: A Focus on End-of-Life

I hereby undertake to carry out my research in such a way that:

- there is no apparent legal objection to the nature or the method of research; and
- the research will not compromise staff or students or the other responsibilities of the University;
- the stated objective will be achieved, and the findings will have a high degree of validity;
- limitations and alternative interpretations will be considered;
- the findings could be subject to peer review and publicly available; and
- I will comply with the conventions of copyright and avoid any practice that would constitute plagiarism.

SIGNED BY	Full name	Signature	Date
Principal Researcher/ Student/External applicant	Clare Josephine Rodseth		**22 Jul 2016**

APPLICATION APPROVED BY	Full name	Signature	Date
Supervisor (where applicable)	Prof Harro von Blottnitz	Harro von Blottnitz	28-07-2016
HOD (or delegated nominee) Final authority for all applicants who have answered NO to all questions in Section1; and for all Undergraduateresearch (Including Honours).			
Chair : Faculty EIR Committee For applicants other than undergraduate students who have answered YES to any of the above questions.	G. S; Hole		13/09/16

Page **1** of **1**